INTERNATIONAL ASSOCIATION OF HYDROGEOLOGISTS SELECTED PAPERS

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Groundwater and Human Development

Editors: Emilia Bocanegra
Mario Hernández
Eduardo Usunoff

GROUNDWATER AND HUMAN DEVELOPMENT

SELECTED PAPERS ON HYDROGEOLOGY

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Groundwater and Human Development

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Preface

The survival and well-being of humans are primarily related to their access to water resources, both in meeting their basic biological needs and as the major drive for socio-economic development. In essence, water has been part of human culture (in its widest interpretation) since the appearance of human life on Earth.

The development of groundwater resources to meet such needs is a relatively recent component in the history of humankind, mainly because the technology needed for their exploitation is more complex than that required for surface water resources. However, in contrast to surface water, the overwhelming availability of groundwater resources globally, which are also naturally better protected against contamination, clearly indicates that the future of humanity is intimately linked to the quantity and quality of the world's groundwater resources.

It should be pointed out that the sixth phase of UNESCO's International Hydrological Program highlights the objectives of the ongoing plan: 'In continuation, IHP-VI (2002–2007) is based on the fundamental principle that fresh water is as essential to sustainable development as it is to life and that water, beyond its geophysical, chemical, biological function in the hydrological cycle, has social, economic and environmental values that are inter-linked and mutually supportive.'

Bearing that in mind, the IAH (International Association of Hydrogeologists) and the ALHSUD (Latin-American Association of Groundwater Hydrology for Development) agreed to organize a joint Congress on Groundwater and Human Development, held from 21–25 October, 2002 in Mar del Plata, Argentina. The following topics guided the technical sessions and other related activities.

- Groundwater and Quality of Life
- Groundwater in Urban, Suburban and Rural Systems
- Transboundary Aquifers
- Hydrogeology of Large Plains
- Coastal Aquifers
- Methods for Groundwater Studies
- Education in Groundwater
- Groundwater Management

This book contains selected papers as a way of showing the contribution of modern hydrogeology in addressing the ever-increasing need of human beings to meet their needs for fresh and safe water. Benchmark conference papers by well-known experts have also been added, as well as material from some of the workshops.

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Foreword

Groundwater is the essential component of the hydrological cycle which facilitates the unique behaviour of water on the continents: water continues to flow on the land surface and sustain large plant communities during the periods without rainfall. This is due to the continuous supply of slow-moving and long turnover time groundwater. This is well known scientifically, but in practice it is not duly recognized by many engineers, policymakers and social leaders, whose attention typically focuses on the visible surface water, ignoring the hidden to sight, underground diffuse water flow. The consequence has been a form of freshwater resources development and management, and even of Nature protection, which is highly biased since it uses and takes advantage of only a part of the water cycle. Some exceptions are found in arid and semiarid lands, where the scarcity and irregularity of surface water has in the past fostered the primitive but ingenious and challenging development of relatively small quantities of groundwater needed for urban supply and socially-complex agricultural communities. These works are mentioned in Babylonian and Summerian documents, and in the Bible, and many of them are still operating from Iran to Spain and Morocco.

Until the mid-19th century basic scientific knowledge of groundwater was virtually absent and even erroneous concepts were applied, some of them deriving from classical times. Moreover, it was relatively costly and difficult to reach and abstract groundwater. Gravity drainage was needed, since only very limited and low efficiency human and animal energy for water abstraction from wells was available. But the situation changed dramatically at the end of the 19th century, and was like a revolution in the mid-20th century. This was due to the more or less simultaneous development of the turbine pump, the widespread availability of relatively cheap energy, and the easy mechanical drilling of deep tube-wells by a large diversity of rigs adapted to a wide range of local technical capabilities.

This means that instead of building every time larger and more costly surface water works and transfers, the aquifers – which have a wide geographic distribution – can be easily developed for groundwater abstraction, and even thick, low permeability formations can support relatively high yielding wells.

Although groundwater development can be carried out under the sponsorship of public entities within large water supply projects, most of it has been and is made by private owners and small communities, using their own means or by applying relatively small money loans. This leads to developments outside the main official investment projects, although low rate loans and subsidies can be provided by official organizations. The result is what Prof. Llamas has called a bottom up "silent revolution" of immense proportion.

This has allowed significant improvements in fresh water availability in developed and industrialized areas, but the most dramatic influence is in poor countries where, for the first time, good quality drinking water is becoming available and global health conditions are greatly improved. Agriculture and animal rising activities in poor areas, often traditional famine areas, have also been gradually transformed into human settlements which can not only feed their people but even produce a surplus for selling. This has been the drastic change in many regions of India and China, and is the key for development in countries like Mexico, Peru and Bolivia. It is also the hope for deprived areas of Africa when aquifers are able to yield enough water in a sustainable way, which often they do even in hard rock areas in dry regions.

There are several negative sides to these highly positive aspects. The most conspicuous negative side effects refer to extensive groundwater level drawdown when exploitation is intensive. This can increase abstraction costs, force the modification of well depths and make the energy supply systems and pumps inadequate. In extreme cases the exploited aquifer cannot yield enough groundwater due to temporary or permanent early depletion of reserves. This is a serious concern for groundwater users and for the sustainability of the development, but happens when the reference used for evaluating the performance is the starting situation of groundwater levels and well depths, and the initial water quality. However, progressive drawdown when groundwater development increases is a natural consequence of aquifer properties; this drawdown is needed to direct recharge to the abstraction points instead of going towards the natural outflow areas. If actual recharge – often greater than natural recharge – is not exceeded, a new steady-state can be obtained, but in many cases it is attained after a long transient period of years and even many decades, which may result in some cases in relatively large drawdowns. To non-specialists it may appear that recharge is exceeded, when it is not. To counterbalance this intrinsic effect, progressively increasing hydraulic and economic efficiency are needed, within the bounds of rational behaviour, to be able to afford the higher water costs. In other words, the reference state to evaluate performance should be not the initial one, but the final one or an evolving one for long transient stages. This is seldom done, and is the source of many unfounded concerns. But this is not easily understood and is outside common experience. Education and an institutional framework is needed.

Groundwater quality deterioration is another possible negative side effect of aquifer development. The causes are diverse: seawater intrusion in coastal aquifers; upcoming of deep-seated saline waters; abstraction of water with undesirable natural constituents such as F, As, Fe, Mn; enhanced and induced infiltration of used waters and irrigation return flows which may be saline, polluted with nitrates, or containing some undesirable chemicals and organic substances, including mineral oils and derivatives from human activities. Counter-measures are diverse and often not easy to apply. Prevention is the most effective and often the only sound policy. A big loser in intensive aquifer development is Nature: many habitats, rivers, springs and wetlands may be seriously damaged or may even disappear if effective conservation measures are not introduced.

All this means that there is a limit to groundwater development in a region if the often enormous initial benefits are to be maintained and "Nature rights" have to be taken into account and preserved, up to a reasonable level which has to be socially acceptable. Humanity cannot live without a healthy environment. It is normal that once the survival of local people is assured, the protection and preservation of Nature has to be considered as an essential part of social development.

The role of human science, technology and management capabilities is to preserve the positive results while controlling and correcting the negative consequences, and in any case looking for a sustainable net social benefit.

The limit to groundwater development is not a fixed quantity but varies with circumstances and priorities, and over time, and depends on scientific and technological progress. In the same way sustainability is a variable goal and not a fixed value. This means flexible and adaptable policies, and restoration when it is needed.

Groundwater development is carried out dominantly by individuals and small communities. For them the existence of some limit to regional development is not generally appreciated and accepted; furthermore, measures towards sustainability cannot be carried out individually but through the agreement of all aquifer users. Obtaining this agreement is the task of institutions and social forces. This is the way to circumvent the "tragedy of common goods". Needed, therefore, are a general understanding of the common problems, their solutions, and the feasibility of attaining them in practice. Also needed is an institution capable of setting rules and empowered to apply them, and to penalize those who break them. This institution must be free of vested interests, political manoeuvring and corrupt practices, and be able to eradicate perverse subsidies. However, rules cannot be effectively applied and accepted if groundwater users – stakeholders – do not effectively participate in management, and share some responsibility through adequate users associations. These are the three main and necessary aspects of groundwater intensive use sustainability for poverty alleviation and quality of life improvement.

Hydrogeology is a relatively new science and technology, and consequently many aspects are still not adequately known and experienced. This means that in some cases progress is a trial and error process, with successes and failures, but walking in a definite positive direction. Some of what are considered recent "big failures" in groundwater use, that have been widely broadcast by the media as such, are mostly sensationally presented and lack well-documented and measured facts. This may seriously damage the essential role that groundwater must play in water resources development. Such is the excessive publicity against intensive use of groundwater in China and India, and even in Spain, playing down the true causes of water problems: the damaging effect of mismanagement, the lack of will by many governments, and the ill-conceived lobbying to foster expensive, large water works. It is a significant failure of the 3rd World Water Forum Ministerial Declaration, in Kyoto, March 2003, to include any reference to the demonstrated major benefits of groundwater to mankind for solving serious water supply and health problems, and to foster poverty alleviation through increased food production. This in spite of the considerable effort made by several aware organizations to show the facts and realities.

The widespread presence of excessive arsenic levels in water supply wells in Bangladesh and West Bengal, and in other South Asia countries and in Argentina, is also presented by some individuals and agencies as a major failure, and sometimes as a mass poisoning of the population. Things can be viewed otherwise. In the past, Bangladesh was a place of frequent water-borne diseases, with high child mortality and a large proportion of disabled population. Groundwater brought a solution by providing clean and pathogen-free water. Some millions of mostly privately owned tube wells are now supplying pathogen-free water to a large fraction of its approximately 130 million inhabitants. The result has been that general health conditions improved dramatically. At that time, however, it was not suspected that a large fraction of groundwater wells had or were to develop high to very high natural arsenic contents. Over the past 20 years this exposure has affected the health of about 2 million persons, with the death rate by different arsenic related cancers being about 3000 persons per year; a dramatic health situation, even if not as bad as it was before. This has prompted a well known and fully understandable reaction, that has been used in some cases to discredit groundwater as a reliable water resource capable of providing cheap potable water, thus improving health conditions and also contributing to alleviate poverty. When this is used to favour other solutions, mostly costly and environmentally unfriendly large surface water projects, social damage can be worse in the long-term.

In Bangladesh groundwater is still a feasible solution to urban and rural water supply, but affected wells have to be replaced by new, often deeper, properly constructed wells in which the prone-to-arsenic contamination upper layers need to be isolated and redox conditions in the deeper layers are preserved. The use of existing and new As–free wells has to be for drinking purposes. This can be achieved by reorganising the supply systems, or alternatively abstracted groundwater has to be treated before distribution with the treatment wastes duly collected and disposed of. This is expensive for a poor population and in some cases unaffordable for individuals and small communities. It is not unaffordable for governments and international aid, however, especially when the proposed alternative solutions are undertakings which are more expensive.

The lesson to be learned is that solutions have to be evaluated not only by their direct costs, but also by the indirect costs that may evolve, though some of the latter may be neither apparent nor foreseeable when the development starts. Overall, groundwater is and will be a reliable water resource for human development which will help to solve water supply issues, improve health conditions and alleviate poverty. But it is not free of secondary effects. This means that development which is often dominated by action from the roots needs institutional control and tuition, with assessment by groundwater experts and the careful monitoring and study of data. This also means a continued research and study effort, and the permanent transfer of knowledge in order that the economically weakest can receive the benefits of scientific and technological advances. Furthermore, the mass media should continue its important task of inquiring and making known what is wrong, but without distorting the reality, in order to not abandon or neglect what is a reasonable, effective, cheap and environmentally friendly solution, in spite of some drawbacks and failures.

Promoting groundwater for human development and enhancing its capacity to alleviate poverty and improving quality of life is one of the leading roles of the International Association of Hydrogeologists (IAH), as well as promoting scientific and technological transfer. This is why the Congress of Mar del Plata (2002) was especially welcome, and this publication of Selected Papers is considered an important one. The IAH is very grateful to the Congress Organizers and to the book editors for their considerable efforts and for providing the opportunity to present a sample of world-wide groundwater problems and solutions, with a flavour of what happens in Latin America. Special mention is due to the continuous efforts of Eng. Emilia Bocanegra, Dr. Eduardo Usunoff, Dr. Daniel Martínez, Dr. Hector Massone and Dr. Luis Vives, as visible representatives of a large group of enthusiastic collaborators.

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Mar del Plata Statement on Groundwater and Human Development

XXXII IAH & VI ALHSUD
Congress 2002

Over 400 participants from more than 40 countries gathered at the Congress on Groundwater and Human Development in Mar del Plata, Argentina, 20–25 October, 2002

In recognizing that:

- water, in addition to being a natural resource that is vital for sustaining any kind of life and human development, embodies important social, ecological, aesthetic, cultural and religious values;
- true human development is achieved when spiritual and cultural values take precedence over purely economic considerations;
- access to a safe drinking water supply and adequate food is essential for all individuals;
- the United Nations Millennium Statement, 2000, established as an objective the eradication of extreme poverty suffered by one out of six human beings, and that this goal can be achieved by 2015 if access to a safe water supply and sufficient food can be guaranteed for all.

The Organizers, after wide consultation with the participants of the XXXII International Congress of IAH (International Association of Hydrogeologists) and the VI Congress of ALHSUD (Latin American Association of Groundwater Hydrology for Development), which met in Mar del Plata, Argentina, 20–25 October, 2002

***state that*:** rational development of groundwater is the most economic and rapid means of providing potable water for most regions of the world, particularly in poor rural areas, and groundwater represents the most readily accessible resource for maintaining small irrigated areas that can alleviate the scarcity of food and the effects of famines caused by droughts;

***agree upon the fact that*:** data from the United Nations presented at the Fresh Water Conference (Bonn, December, 2001), the World Summit on Sustainable Development (Johannesburg, September, 2002) and similar international events suggest that small donations or soft credits can solve the lack of safe water within a decade provided that the funds are duly channelled and effectively and efficiently distributed;

***remind that*:** in seeking for solutions to water problems that inhibit human development, alternative methods aimed at integrated water resources use should be considered. In this context it is worth mentioning that, among other arguments and in contrast to the large capital expenditure and excessive time required for the completion of major dam projects, groundwater development is fast, does not require a large initial financial investment, delivers a better service to the people, steadily improves social welfare and is significantly less prone to corrupt practices;

***draw attention to the fact that*:** uncontrolled and ill-planned use of groundwater is counter-productive, preventing an adequate and rational use of the resource; this highlights the need for good education and information, and the involvement of all stakeholders, making them feel to be an active part of aquifer management and the protection of fresh water resources from contamination and lack of safety, resulting from recent, insidious and barely reversible events;

***state that*:** institutional inertia, legislative deficiencies and the poor performance of many international, national, regional and local organizations must be corrected in order to encourage groundwater development, promote sustained groundwater use, prevent the release of subsidies that can distort the economy and minimize the damaging effects of special interest groups;

***request that*:** the attention of government ministers attending the third WWF be drawn to the fact that there is a serious need to strengthen groundwater-related research and educational programs at national and regional level;

***recommend that*:** national governments, international donor agencies and the International Hydrological Programme of UNESCO, continue supporting groundwater monitoring and data collection, groundwater integration within land use planning, training facilities, institutional capacities development and keep a particular focus on Latin American regions facing extreme poverty;

***ask that*:** within the numerous studies carried out by national and international organizations, most notably the World Water Assessment Programme of the UN, special attention be given to improving knowledge and assessing the total benefits and costs of using groundwater to alleviate poverty.

Section 1

Plenary conferences

CHAPTER 1

Groundwater and human development

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ABSTRACT: Water as a natural resource has important economic and social value. In the last few decades, an outstanding, almost spectacular, increase in the use of groundwater has been achieved in most arid and semi-arid countries. This has come about with very little governmental control. That is the main reason why, together with very important benefits, some drawbacks have been resulted, which have very often been exaggerated by the media. The UN Millennium Declaration has proposed, among its main goals, to halve the number of people without access to potable water or who suffer from malnourishment by 2015. Groundwater development is a keystone for reaching both goals.

INTRODUCTION

It is well known that water is an essential element for the existence of any form of life. In fact, about two-thirds of the human body consists of water, which is renewed every six to eight weeks. Water is also an essential element for many economic undertakings such as irrigated agriculture, which is the way approximately 40 per cent of the food consumed by human beings is produced.

The relevance of water extends well beyond utilitarian examples of this kind. It often assumes an importance that is difficult to quantify. Notions of the intangible value of water can rest on its importance to the environment, as well as its symbolic relevance to most cultures and religions. As a consequence, conflicts over water resources often involve an additional emotional factor.

Some historians use the term 'hydraulic' to describe the first civilizations. These civilizations were born between six and seven thousand years ago in great valleys in large arid regions such as the Nile in Egypt or the Tigris and the Euphrates in Mesopotamia. Nomadic hunters became farmers and began to manage local water resources through small infrastructures. However, building and managing these works required a collective effort, which naturally led to structured societies living in the same 'civis' (towns). This tradition, or even necessity, of managing water resources collectively has continued to the present day.

Just about all major hydraulic works within the last 100 years have been under the financial and operational control of government agencies or institutions. On the other hand, groundwater resources can be developed by means of minor works such as wells and galleries, thus allowing individuals or small communities to take control.

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As a general rule, up until 40–50 years ago, the amount of water yielded by these small infrastructures for urban and farming purposes were hardly significant. However, this has been reversed over the last half century due to technological advances in well drilling and pumping systems, as well as to increasing knowledge about the origin, occurrence and movement of groundwater resources. These three factors have led to a dramatic increase in the use of groundwater resources in arid and semi-arid countries. Perhaps the most outstanding case is India, where groundwater resources have been used for the irrigation of more than 30 million hectares over the last 30 years. As a consequence, this country has moved from constant and generalized periods of famine to become a relevant grain-exporting country. This spectacular development has been inspired, financed and achieved by individuals or small communities, with little planning or control by government agencies (Llamas & Custodio, 2002).

However, this development has given rise to a series of problems, which have been blown out of proportion. Thus large sectors of the population now believe the ‘myth’ that groundwater is a fragile and delicate resource. ‘Every well either dries up or becomes saline’ is a false, albeit widespread, belief. The practical consequence of this is that surface water is often considered to be the only reliable resource, even if competent studies prove local aquifers to be more advantageous from the points of view of the economy and the environment (Custodio, 2000).

A BRIEF CONSIDERATION ON HUMAN DEVELOPMENT

Although this paper is obviously not the best place to comment on human development, it is nevertheless important to emphasize certain aspects in as much as they relate to groundwater.

True human development cannot be equated with ‘having more’, but rather with ‘being more’. Cultural and spiritual values must always take precedence over purely materialistic or hedonistic goods. At the same time, it is universally accepted that in order for this to occur, a human being must have a minimum level of material benefits such as food, drinking water, housing, education and adequate public health care.

The inhabitants of regions whose average annual rent per capita is lower than US$ 500 are considered to be living under extreme poverty. The estimate is that out of the six thousand million people that inhabit the planet, about one thousand live below this poverty threshold. On the other hand, there are approximately one thousand million people, living mainly in developed countries whose average rent per year is over US$ 10,000 (that is, about 20 times higher than people living in poorer countries). Since this comparison, based exclusively on per capita rent, is somewhat simplistic, the United Nations (UN) have been using a more complex criteria, which takes into account factors such as education and life expectancy, for years.

It would be reasonable for the onlooker to question why the ‘rich’ should take an interest in raising the level of the ‘poor’ instead of assuming that they are poor because they work less or are not as intelligent. There are at least two commonly acknowledged ethical reasons for not accepting this argument. First, the intrinsic dignity of all human beings and second, the sense of fraternity, or solidarity, that exists between all people (Llamas, 2001). In addition, other practical (or even selfish) considerations point out that it is not possible to implement global sustainable development while poverty exists. Thus the 2002 World Trade Organization meeting in Doha concluded that it was necessary to reduce the tax barrier on farming products in a clear attempt to help poorer countries. Another reason, which has become more prominent after 11 September, 2001 is the fact that poverty produces an ideal culture for future terrorists.

The role that water plays in the eradication of poverty is essentially twofold. First, it is necessary to provide drinking water, sanitation and hygienic education to less-developed countries in order to reduce the mortality rate. Second, and generally not as important, is the need to implement small-scale irrigation systems so as to help provide enough food for those regions whose inhabitants suffer from chronic malnutrition or find themselves exposed to famine whenever prolonged drought periods occur (Moench, 2002).

In 2000, the UN released the Millennium Declaration, which consisted of an analysis of the general state of affairs on the planet, together with a series of action plans for the future (United Nations, 2000a, 2000b). Two of these actions aim to halve the proportion of the Earth's population that does not have ready access to drinkable water or suffers from malnutrition by the year 2015.

IMPORTANCE OF GROUNDWATER AS A MEANS TO ERADICATE POVERTY

International conferences such as Bonn's International Congress on Freshwater (December, 2001) or the four preparatory sessions for the World Conference on Sustainable Development, also known as Rio +10, (Johannesburgh, August, 2002), have upheld the conclusions of the Millennium Declaration.

The International Conference in Bonn (German Federal Government, 2001) pointed out that the investment needed to provide a basic water supply and water treatment systems in developing countries (whose collective population is about 1,000 million people) will amount to about US$ 20,000 million in the next ten years. Developing countries are to contribute half of this amount, while soft loans and donations should make up the rest.

This overall figure (which might appear high) adds up to a mere US$ 10 per person each year. As stated above, there is a similar number of people in developed countries whose yearly rent per capita is over US$ 10,000. In other words, if each of those people in developed countries donated US$ 10 a year, the problem would be solved in just one decade. This annual donation would constitute less than 0.1 per cent of the average income, less than the amount people in developed countries spend on food for domestic animals or on ice cream.

These data show only the tip of the iceberg of the serious problem that unsustainable consumption constitutes in developed countries. This is barely ever mentioned by neo-Malthusian theorists, who prefer to emphasize the problems, real or fictitious, associated with the increase in the world's population. It is also often forgotten that most developed countries are still far from fulfilling the UN's proposal that they should contribute 0.7 per cent of their gross national product to help developing countries (Llamas, 2002).

Thus it appears clear that solving the problem of poverty for millions of people does not imply an extraordinary effort on the part of rich countries. However, the need for adequate management of these investments adds extra complexity to the problem, as donations are meant to be catalysts to the improvement of the institutional and organizational capabilities of developing countries, rather than being mere alms to the poor. On the other hand, it is a well-known fact that the final destination of some investments has been the purchase of weapons or the personal bank accounts of unscrupulous politicians (Llamas, 2002).

There are a number of reasons why newly implemented drinking water supplies and irrigation systems should be based mainly on groundwater resources. First, groundwater infrastructures are often cheaper than equivalent surface water infrastructures. Second, investments for groundwater schemes can be more easily apportioned over time while yielding results almost from the start; in contrast, hydraulic works based on surface water resources rarely take less than 20 or 30 years to be fully-functional. Third, groundwater supply and irrigation systems are usually smaller-scale, thus allowing for more progressive participation from the beneficiaries. In fact, experience shows that in many countries – take India as the most spectacular example – governments began 20 or 30 years ago by building a modest number of irrigation wells. However, the new technology was soon assimilated by the local farmers, who developed new wells at their own expense and at a much quicker pace than the government. It must be noted, though, that this quicker rate can be excessive at times and must be regulated by the government in order to ensure sustainable and egalitarian exploitation of groundwater resources.

Groundwater irrigation systems present a high degree of insurance against drought. As a consequence, poor farmers, ruined by prolonged periods of drought, have become a rare sight in these areas. Another positive consequence of water supplies is that farmers have the possibility of investing in newer and improved technologies, thus obtaining more efficient harvest returns. In some places, poor farmers have been able to send their children to university, thus setting off slow social change towards the emergence of an educated middle class (Moench, 2002).

MAIN OBSTACLES TO ATTAINING ADEQUATE GROUND WATER RESOURCE MANAGEMENT

As stated above, the use of groundwater has risen dramatically in arid and semi-arid countries, as well as in small islands and around important urban centres. Nevertheless, groundwater resources remain heavily under-used in poorer countries, even though these resources hold the key to development. The following paragraphs provide a succinct explanation of the main obstacles to providing an adequate (neither excessive nor inadequate) use of groundwater resources in both rich and poor countries.

Scientific and technological deficit

Those responsible for water management policies are, in most countries, people who believe in the necessity and excellence of major surface hydraulic structures. In addition, these people often present a serious lack of hydrogeological knowledge, which leads them to an attitude of 'disregarding what they ignore'.

This problem is a serious one, particularly in developing countries, as there is virtually no one prepared to manage water resources of any kind. When confronted with necessity, developing countries often go to large credit institutions or donor countries, which almost exclusively propose the implementation of major surface water projects.

Lack of solidarity (due to lack of information)

Groundwater has been, and still is, taken advantage of as a resource mostly by individuals. Each well functions whenever the owner chooses, without taking into account the effects that might have on the neighbouring beneficiaries of surface or ground water.

While surface water irrigation is usually controlled by one 'boss', or by local governing bodies, aquifer exploitation is often carried out by thousands of individual users who act completely independently from each other. The latter attitude is probably a consequence of a general lack of knowledge about how aquifer systems work.

Newness of collective aquifer management institutions

Institutions for the collective management of pooled resources are not rare these days. A close analysis of how these groups work openly challenges the commonly held perception of 'the tragedy of common property goods'. This refers to different types of natural resources such as hunting or fishing, but there are some well-known examples in the field of surface waters, like the Water Tribunal of Valencia, which has worked well over the last eight centuries. Nevertheless, as stated above, intensive use of groundwater only dates back to one or two generations ago and thus the majority of groundwater control institutions are still developing in most countries (Hernández-Mora & Llamas, 2001).

This is a crucial field, which will probably experience a great deal of activity over the next few decades. The basic idea is that rational aquifer management can be based exclusively neither on a series of laws nor on the establishment of semi-political government bodies entrusted with the control of aquifer abstraction (Llamas et al., 2001). It seems important to establish these organizations in a bottom-up manner, ensuring the necessary help from the relevant government agencies. However, it seems impractical to try to implement this in a very rigid fashion, with a 'big brother'-like figure to decide where every drop of water goes. That could only work in small countries in a permanent state of war.

Institutional inertia

Once again, it seems important to note that the relative novelty of using groundwater as a relevant resource results from the public's and most politicians' belief in paradigms that were pervasive one or two generations ago, but are now obsolete.

In order to accelerate change, it is necessary to launch education campaigns about the effective use of groundwater (Llamas et al., 2001).

Vested interests

Last, but not least, there has been a historical attitude that water should be heavily subsidized and that major hydraulic structures should be built with public funds not by private companies. This situation has helped to consolidate a series of social sectors (surface irrigation farmers, construction companies and others), which now benefit thanks to public subsidies policies justified half a century ago. These groups constitute an obvious nucleus of resistance to change.

Corruption is another unfortunate circumstance that affects most countries, albeit in varying proportions, and which indirectly opposes a turn towards taking advantage of groundwater resources. A proportion of the public funds intended for hydraulic infrastructures may well end up in illegal payments to politicians or public servants (Delli Priscoli & Llamas, 2001). Fortunately, however, the Organization for Economic Co-operation and Developemnt (OECD, 2000) began a campaign against corruption a few years ago. It is expected that the fruits of this interesting initiative will be noticeable reasonably soon (Llamas, 2001).

CONCLUSIONS

The intensive exploitation of groundwater carried out in arid and semi-arid countries over the last few decades has yielded abundant benefits to human beings.

The use of groundwater is a key element in the eradication of poverty and the dramatic reduction of the proportion of the world's inhabitants who still do not have ready access to drinking water and/or suffer from famine.

As a general rule, intensive use of groundwater has been carried out and funded by individuals with no or very little planning on the part of competent hydrologic authorities. As a consequence, a series of problems have arisen. These can and must be solved in order to achieve sustainable development of this important natural resource.

As the 2001 Bonn Congress reminds us, this planet's water resources will only be enough to cater for everyone if the concept of its management changes.

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CHAPTER 2

An overview of groundwater resources in Latin America

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ABSTRACT: An overview of groundwater resources in Latin America is given according to four aspects: natural supply, availability, emerging uses and associated conflicts, and future prospects. Numerical indicators supporting the discussion are also given, mostly for the end of the twentieth century. There are sub-regions where the annual consumption maybe as high as 27,570 m^3/inhabitant, whereas in other areas marked deficits are observed. One of the main problems is the growing conflict (intrinsic and between uses) induced by very large urban settlements and great social inequity. Groundwater resources are gaining increasing importance and they represent an increasing proportion of the water supplies used for different applications. A lack of planning, education, awareness and policies orientated towards agreed water use and preservation jeopardizes the sustainability of aquifer exploitation and creates one of the more serious problems for the region in the medium and long term.

INTRODUCTION

In this attempt to present a succinct overview of the current panorama of groundwater resources in Latin America by looking at the internal causes, four analysis axes are presented: the supply of natural waters in the region, the availability of groundwater, the current utilization status and the associated conflicts, and future projections. This work discusses these four axes because, even though they are clearly different in concept, they work together as a whole to present a complex overview.

WATER NATURAL SUPPLY

Simple observation of the distribution of rainfall around the planet shows that South America, Central America and the Caribbean have a privileged amount/surface ratio, which is even greater than Southeast Asia and Central-Guinean Africa.

More than three-quarters of the territory of approximately 19,880,000 km^2 have precipitations between 500 and 1,000 mm/year, and more than half of the surface receives rainfall of between 1,000 and 2,000 mm/year.

It should also be pointed out that losses caused by evapo-transpiration are also high and that there are large arid or semi-arid sub-regions (two-thirds of Argentina, northern Chile and the Pacific coast up to Ecuador, northern Mexico, and northeastern Brazil). As a whole supplies from the atmospheric phase of the hydrological cycle can be considered generous.

Latin America contains large basins with fluvial modules of more than 20,000 km^3/sec (the Amazon, Orinoco-Apure and Paraná-Uruguay), vast humid areas reflecting the relative importance of the surface terrestrial phase of the cycle and groundwater resources (in various locations and with different degrees of potential) that can reach very significant levels, such as the Guaraní Aquifer (Argentina, Brazil, Paraguay, Uruguay), which covers some 1,195,000 km^2.

If freshwater supplies are considered as a whole, Latin America has plenty to offer: around 27,570 km^3 per capita (an order of magnitude above that of Spain for example), with peak values of 61,750 km^3 per capita in Paraguay and 1,642 km^3 per capita in Perú. Exploitation by the end of 2000 was above 260 km^3, from 78 km^3 in Mexico to less than 1 km^3 in Paraguay, the latter being the country with the highest unused available water rate. (Fernández Cirelli, 2001; UNSD, 1999).

With the exception of México, Cuba and the Dominican Republic, the ratio between water exploitation and water availability is estimated to be lower than 10 per cent, a situation that is not expected to present variations until at least 2025 (UNSD, 1999).

Some 71.1 per cent of the used volumes are destined to agricultural activities, with a maximum of 92 per cent in Uruguay and Honduras and a minimum of 37 per cent in Colombia, which is the only country where this application is not the most important one.

Domestic use represents 20.5 per cent of the total, with a maximum in Colombia where it represents 59 per cent and a minimum in Honduras with a bare 4 per cent. An average of 84 per cent of the population has access to drinking water, with a maximum participation in Uruguay and Panama (99 per cent) and a minimum in Paraguay (70 per cent). Only 41 per cent of the rural population has access to drinking water, with extremes that vary from 85 per cent in Cuba to barely 6 per cent in Paraguay (Fernández Cirelli, 2001; UNEP, 2000).

Industrial activities (including mining) use the remaining 8.4 per cent, with Brazil, Bolivia and El Salvador using above 20 per cent and the Dominican Republic and Cuba not reaching 1 per cent.

GROUNDWATER AVAILABILITY

An exercise by Hernández (1990), based on Lvovich methodology (1975) and UNESCO (1971), which can be useful for global and territorial balances, comparing the total emerged lands of South America with global figures is shown on Table 1.

Table 1. Water balance for South America.

	Global	South America	%
1. Surface ($km^2 \cdot 10^6$)	132,3	17,8	13.4
2. Precipitation (km^3)	109,305	29,355	26.8
3. Evapotranspiration (km^3)	71,765	18,975	26.4
4. Surplus (2–3) (km^3)	37,540	10,380	27.6
5. Edaphic humidity (km^3)	83,360	22,715	27.2
6. Fluvial discharge (km^3)	11,995	6,640	55.4
7. Groundwater discharge (km^3)	25,545	3,740	14.6

Indicators 2–7 were calculated as hypothetical water height and then the effect of the surfaces involved was taken into account in order to obtain a volumetric result. Two factors should be noted: on the one hand, the correspondence between surfaces and groundwater discharges, given the proportionality of effluence perimeters with the areas included; and on the other hand, the influence of the great South American rivers (e.g. the Amazon, with a module of 220,000 km^3/sec, i.e. 16 per cent of the world's fluvial discharge).

If we assume that this regime could be constant for a 50-year period, groundwater discharge would be approximately equivalent to the recharge, leaving aside the basic volume that flows to the sea and that has been computed in the fluvial discharge. The balance would then show the importance of the renewal of groundwater in most of the region.

The occurrence of groundwater includes every possibility: porous, fissured and karstic media. In South America there is a predominance of porous over fissured media and in Central America and the Caribbean porous media predominants over karstic (UNESCO-PHI, 1996).

References to usable reserves in Latin America are not enough to provide accurate figures, particularly in relation to aquifers in fissured media. However, it is certain that to date they are more important than in other regions of the world. Estimates for GWP (2000) give South America a reserve in the order of 30,000,000 km^3; if this figure is added to the estimates corresponding to the rest of the region (mostly in Mexico), the final figure would go up to more than 30,500,000 km^3.

As an example, the estimate for the Guaraní Aquifer (GEF-WB-OAS, 2001), underlying Argentina, Brazil, Paraguay and Uruguay, is around 50,000 km^3, above the average global availability of 12,500 km^3/year noted by the UNDS (1999) and a bit less than half the total groundwater reserves of Brazil (GWP, 2000).

Unlike the situation with surface water, the groundwater reserves in the region increase at the same time as the existing resources are exploited, largely due to the discovery of new sources.

Naturally, due to basic hydro-climatic reasons the greatest groundwater reserves are to be found precisely where surface resources are also abundant. However, in areas with arid or semi-arid climate the occurrence of fresh groundwater provides support for development, for example, in the central-western and southern areas of Argentina, northern Mexico, the Pacific coast between parallels 4° S and 32° S and northeastern Brazil.

The region undoubtedly presents great contrasts, not only in climatic, physical, biotic and hydrologic terms, but also on cultural and socio-economic grounds. The existence of groundwater and its capacity for renewal follow the same pattern, so that the conditions in any given country are usually very diverse.

CURRENT USE OF GROUNDWATER AND ITS CONFLICTS

An analysis of the current use of groundwater resources and the associated conflicts also shows a manifest heterogeneity in the different sub-regions and countries.

Historically, the first exploitations were intended for human use, in the light of the first signs of biogenic contamination of surface waters, as in the case of Buenos Aires City. Industrial use of groundwater started in the 1930s and agricultural use only in the last few decades.

The volumes used for irrigation quickly increased to more than those for the other two applications as a result of increased agricultural activity beyond the margins of the rivers.

According to González (2000), of the 2,766 km^3/year used in Chile, 48.4 per cent was destined for agriculture, 35 per cent for domestic use and 16.6 per cent for industry. In general, it can be said that the current tendency in the region is towards the average use valid for the planet, with a predominance of agricultural applications.

However, there are asymmetries originating from the degree of development of the different countries: whereas industrial use is growing in Mexico and Brazil, the economic crisis in Argentina has caused a recent decline in industrial use.

Naturally, an increase in the use of groundwater resources introduces a number of conflicts and makes the lack of public policies to provide solutions apparent. These conflicts can be either of an intrinsic nature or between different types of use.

The most notorious intrinsic conflict is caused by the imbalance between the provision of fresh running water and sewage services. In the peripheries of large and medium cities, the speed at which fresh running water is provided is faster than the growth of sewage connections and the installation of purifying plants for residual liquids, so the biogenic pollution generated by domestic use causes large volumes of water to become useless.

Only just over 55 per cent of the population has organized sewage systems; *in situ* sanitation for the aquifers, which are the main receptors of this type of pollution, is therefore very important.

If we take into account that, in addition to public supply, individual use is mostly at the expense of groundwater and occurs particularly in highly populated areas lacking sanitation mechanisms, the intrinsic impact of home use acquires alarming dimensions.

Regarding conflicts between different applications of the resource, agricultural activities make use of agricultural chemicals associated with irrigation practices, which has an impact on domestic use. Industrial pollution makes its contribution too, and in turn is affected by the pollution caused by home and agricultural use (in the case of food and pharmaceutical industries).

In the case of agricultural activities, the abusive use of nitrogen-based fertilizers causes an increase in the concentration of nitrates, which due to their longevity in groundwater and because of the lack of urban sanitation constitute a serious widespread problem.

The lack of planning in the distribution of exploitation wells in turn produces physical deterioration of the reserves and the generation of vast depressed areas, a situation sometimes made even more complex by the infiltration of saline waters.

In other circumstances the substitution of a groundwater source with a surface resource – usually due to salinization or pollution – generates a recovery of piezometric levels that affects an infrastructure built during the depletion state, as in the case of Greater Buenos Aires (Hernández & González, 2000).

Unfortunately, in the case of Latin America there are, in addition to all these qualitative and quantitative conflicts, additional complications caused by political, cultural and socio-economic situations. It is well known that the countries of this region have a tendency towards the formation of large metropolitan areas, with 75.1 per cent of inhabitants living in cities and their surrounding areas, and an immeasurable growth of large urban areas (e.g. São Paulo, Mexico City, Lima-Callao, Buenos Aires, Caracas, Rio de Janeiro) where the lack of water is not the only source of conflict.

Every imaginable kind of conflict contributes to these circumstances, which tend to produce a serious deterioration in the chances reversibility and socio-sanitary repercussions as shown by the statistics of the Pan-American Health Organization (Organización Panamericana de la Salud, OPS) and the World Health Organization (WHO).

At the same time, the region has fallen noticeably behind developed countries in such basic issues as education (both formal and informal) about water use, knowledge of the characteristics of groundwater resources, efficient public policies, the presence of the government as a controlling and regulatory entity of the social use of water, together with a growing pauperization of the population and a lack of user participation in the management of resources.

The wave of privatization/concession of the supply and administration of public services in Latin America – more often than not with groundwater sources – is bringing about a tendency towards considering economic factors rather than those inherent to water resources as a social asset.

Thus, the supply of the service is granted to large concentrations of consumers or sectors that produce large amounts of money before it is granted to marginal areas with few economic resources, which are more in need of access to safe water because of their greater exposure to environmental threats.

FUTURE PERSPECTIVES

The actual facts, carefully observed from the realities in the region, do not indicate an encouraging future in relation to important changes to work towards the preservation and rational use of groundwater, despite its being the most secure source of water.

If the relevant states, as usually happens, do not retain the power to establish appropriate policies for the administration of the resource, a joint, harmonic and planned use of surface and groundwater resources is inconceivable. Uncontrolled concessions will only ensure economic profit, the natural goal inherent to any company.

It should be taken into account, however, that there is one universal characteristic shared by groundwater resources all over the world – a their 'invisibility'. As said by Antoine de Saint Exupery through his Little Prince, 'what is essential is invisible to the eye' and this is precisely the case with groundwater. The 'invisibility' of groundwater works against the efforts of geohydrologists to protect them. They are not attractive as a political-electoral objective, it is hard to make politicians and decision makers understand the need for public investment in studies and surveys, and the general population is not very aware of the need of taking care of groundwater, therefore being little involved in its administration, which is not a priority even in scientific and technological research systems.

Within this universal truth, Latin American countries lag far behind. Except countries such as Brazil (with the creation of the Agência Nacional de Águas, ANA), Chile, Mexico and Cuba, the others have either not improved their situations much or have even gone backwards, as in the case of Argentina, where major evaluation studies used to be carried out on a regular basis until the end of the 1980s.

One encouraging example is the Project for Environmental Protection and Sustained Management of the Guaraní Aquifer System, approved by the GEF (Global Environmental Facilities) in 2001. Propelled by the universities of the region and facilitated by the governments of Argentina, Brazil, Paraguay and Uruguay, the project initially faced government skepticism about the need for it in a region characterized by an abundance of surface resources. Widespread protection of the environment is undoubtedly a positive sign.

The road to walk is neither short nor easy. There is still much to be done to equal the achievements of developed countries; the obstacles to overcome before starting a new stage are numerous, diverse and even perverse.

A general consciousness about this issue is not to be expected in Latin America, inasmuch as a great proportion of the population faces other issues such as hunger, poverty, unemployment and social exclusion, which are unfortunately growing worse.

Access to safe water is one more right that must be added to the list of basic human rights which, for our people, acquires the characteristics of a mortgage that weighs upon the future. It is the duty of all geologists, members of associations such as the IAH, ALHSUD, IAEH, ABAS, and many others to avoid the situation in which groundwater resources – the guarantee of a sustainable future – become part of that mortgage. In taking on this mission, they need to be willing to face the day-to-day difficulties that, unfortunately, are numerous in Latin America as they come along.

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CHAPTER 3

Coastal aquifers as important natural hydrogeological structures

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ABSTRACT: Coastal aquifers share many hydrogeological characteristics with continental aquifers. The main difference is the risk of water quality deterioration by salinity increase. This is due not only to natural or induced mixing with present sea water but also to the possible existence of old marine water in deep aquifers and aquitards, and to the generation of saline waters and brines in flat areas at an elevation close to that of current sea level. The mixing of freshwater with 3–4 per cent of sea water is enough to make the freshwater unfit for most uses and may seriously reduce its environmental role. The principles governing the distribution of fresh and saline water bodies in the ground and the mixing mechanisms are reasonably well known, although their evaluation and monitoring require networks and methods of study that are often more complex and costly than those commonly used in continental aquifers. Currently it is possible to devise coastal aquifer exploitation plans to limit and correct salinization problems by applying technological methods as well as institutional management with the effective participation of stakeholders. Coastal aquifers are highly valuable as a freshwater resource and as a regulating, emergency and strategic water reserve, since they occur in the lower reaches of river basins. These areas are often flat, with little opportunity to store surface water, and often where population, economic activity and tourism concentrate. The widely diffused 'hydromyth' that developing a coastal aquifer necessarily means salinization problems is false in most cases. Improved knowledge, case studies and social communication are needed to counter this fallacy and to beneficially use opportunities that are technically, economically and socially of high value.

INTRODUCTION

Coastal areas of continents and islands, including small islands, sustain important human activity. They provide flat land to settle and cultivate crops and are key points for maritime communication; they also tap into an ancestral preference for mild maritime climates and environments, now compounded with tourism and third-age residential areas. Their occurrence at the lower reaches of major river basins with long coastal spans without significant streams or around islands with little permanent surface water makes groundwater in coastal aquifers an important freshwater resource for economic activity (e.g. irrigation, tourism, urban services, industry) and especially for drinking purposes.

A series of circumstances generate the risk of groundwater quality degradation in coastal aquifers by introducing an excess of dissolved salts. Sea water is the most important but not the only source of salinity. Mixing groundwater with only 2 per cent sea water produces a noticeable deterioration. If the proportion is 4 per cent there is a serious impairment for many uses. If it is 6 per cent, the water is almost unusable except for cooling and flushing purposes.

When there is poor management or overuse, some serious salinization problems appear, even if they are localized. This means that quality of life is impaired, economic activity is jeopardized and irrigated agriculture may be seriously damaged.

Salinity is caused not only by present sea water intrusion. These other causes have to be identified for the correct management of the situation, otherwise serious errors will arise. The possible sources of salinity in coastal aquifers are:

- encroachment of modern sea water
- mixing with unflushed old marine water in very slow flowing aquifers or in aquitards
- sea water spray on windy coastal strips
- intense evapo-concentration of surface and phreatic water in dry climates
- intense evaporation of outflowing groundwater in discharge areas and wetlands
- dissolution of evaporite salts from geological formations
- displacement of saline groundwater contained in some deep formations
- infiltration of saline return irrigation flows
- pollution by saline water derived from:
  - mine drainage and tip leaching, especially in salt and potash mines
  - leakage from industrial processes and cooling facilities using brackish or saline water
  - effluents from softening, de-ionization and desalination plants
  - dissolution of road de-icing salt
  - intense evaporation of water from factories
- saline water imported from other areas.

Coastal aquifers can be developed to yield fresh water if groundwater flow characteristics are well known, wells are correctly constructed and positioned, a capable management organization exists, adequate policies have been established and there is the will to protect, conserve and restore the aquifer. Popular support and participation, as well as public education, are important for governance and management.

It has been possible to quantify freshwater–saltwater relationships in coastal aquifers since Badon Ghijben (1889) in the Netherlands and Hertzberg (1901) in northern Germany formulated the pressure equilibrium balance of the freshwater–saltwater interface known as the Ghijben-Hertzberg (G-H) principle. Real situations may differ greatly from the very simplified conditions for which the principle was developed, but even in such cases it is useful to describe and quantify the actual behaviour, if the information is correctly applied. This means that the head of salt water in the ground has to be considered, according to Hubbert's principle (1940) of pressure equilibrium of both fluids at each side of an interface, especially during transient situations or when saline water is being pumped out directly or mixed with freshwater.

The basics and some relevant examples can be found in Van Dam (1997), Reilly & Goodman (1985), Custodio & Llamas (1983), Custodio & Bruggeman (1987) and Falkland & Custodio (1992), as well as in texts such as Todd (1959), De Wiest (1965), Bear (1979) and Strack (1989). Although saline water intrusion is an important hydrogeological issue, comprehensive books dealing with the topic in depth are relatively rare.

Specific contributions can be found in the SWIM's (Salt Water Intrusion Meeting) proceedings. A selection of the best papers from the first ten meetings was prepared by De Breuck (1989). Afterwards biennial meetings have followed in Gdansk, Barcelona, Cagliari, Malmö, Miedzyzdroje (Poland) and Delft, from which proceedings are available, and a new meeting is scheduled in Cartagena, Spain in 2004.

COASTAL AQUIFER CHARACTERISTICS

The behaviour of coastal aquifers, besides the geological characteristics that may derive from sedimentation processes in an interfacial environment, is conditioned by the fixed hydraulic head imposed by the sea and the greater density of sea water.

In most coastal aquifer systems groundwater flows naturally towards the sea driven by the head potential created by inland recharge. Since mean sea water level is practically constant there is no induced flow in it, except for short-range, periodical tidal fluctuations. The equilibrium conditions can be described by the G-H principle when a sharp interface separates freshwater and sea water. The interface depth is α times the freshwater head, both referred to the local mean sea water elevation. α measures the specific weight (γ) difference between salt- (s) and fresh- (f) water: $\alpha = \gamma_f/(\gamma_s - \gamma_f)$, and its value is approximately 40 under normal circumstances (see Custodio & Bruggeman,1987 for more detailed considerations).

Freshwater flow influences salinity stratification. The resulting isocones (isoconcentration surfaces) start near the coastline and plunge into the ground down to the lower boundary of the aquifer. This produces the classic saltwater wedge or the floating freshwater lens in small, thick permeable islands.

The actual situation is more complex and has to be described in terms of three-dimensional heads (Hubbert, 1940; Luszczynsky, 1961) as indicated in Figure 1. Diffusion, and especially groundwater flow-induced dispersion, tends to mix fresh- and saltwater, and this is enhanced by aquifer heterogeneities. But freshwater flow, especially in the zone around the expected position of the interface, drags the mixed water along with it towards the coast (Figure 2). Thus, this mechanism helps in limiting the thickness of the freshwater–saltwater mixing (transition) zone and also induces some saltwater flow towards it to keep the salinity balance. The result is that there is a mixing zone, the thickness of which depends on aquifer circumstances, and which can vary from a close-to-sharp interface to a very wide zone. The resulting mean sea water head in the ground is slightly lower than mean sea water level. At a given moment the sea tide induces oscillations and the consequent fluctuation of head and instantaneous groundwater flow velocities (Cooper et al., 1964). The same happens for the seasonal cycle of groundwater recharge (see details in Custodio & Bruggeman, 1987). All this means that in real aquifers circumstances may be complex and their quantitative description needs an elaborate and costly monitoring network that considers the three-dimensional flow and salinity pattern.

In confined aquifers groundwater discharges into the deep sea bed directly or through permeable cover materials if the freshwater head is enough to compensate for the denser sea water column above, as shown in Figure 3. Otherwise there is no discharge.

Near the coast or at the sub-marine outflow of a confined aquifer, regional vertical groundwater flow components are important. When considering aquifer layering and

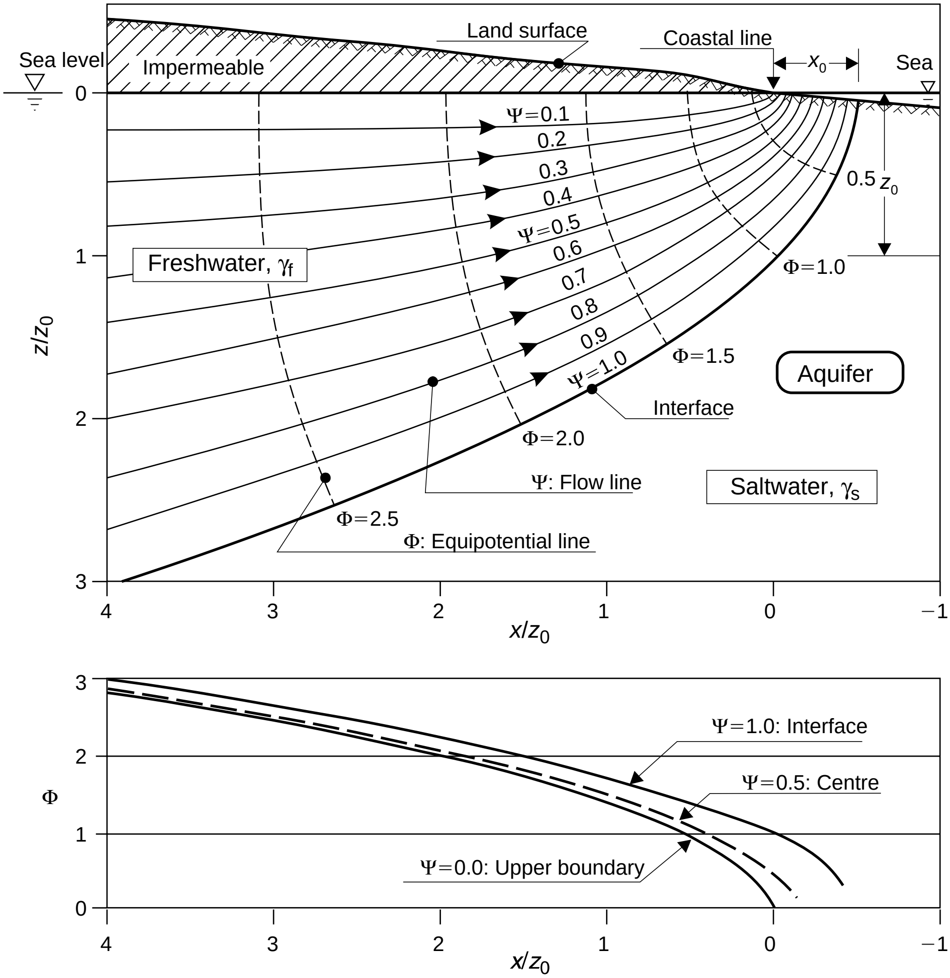

Figure 1. Glover's (Cooper et al., 1964) 2-D exact solution for a thick, homogeneous coastal aquifer with freshwater flowing on a sharp interface with stationary seawater. It is a homogeneous, isotropic aquifer of infinite thickness, confined at sea level. The outflow face is horizontal at sea level. Total freshwater flow is q_0, divided into 10 flow tubes. Freshwater potential is $h = \phi z_0/a = \phi q_0/k$; $\alpha = \gamma_f/(\gamma_s - \gamma_f)$; γ = specific weight of f = freshwater, s = saltwater, ϕ = equipotential number, z_0 = depth below sea level of the interface at the coast line, $x_0 = z_0/2$ = width of the freshwater outflow face in the sea, x distance to the coast line, z = depth below sea level. The lower part shows freshwater head profiles along the interface, the top of the aquifer and at an intermediate position. The interface depth corresponds to the G-H principle applied to heads along $\Psi = 1.0$, but it is deeper if heads are measured along $\Psi = 0.5$ or 0.0.

heterogeneities, these vertical flows translate into different heads in the various aquifers and sub-aquifers. This implies different freshwater–saltwater relationships. In some layers there is no flow when the freshwater head is not enough to balance the heavier marine water column at the outflow face (Figure 4). Upstream freshwater head control may have a decisive influence. Figure 5 shows the same confined aquifer structure under two different circumstances, one with high upstream freshwater head (freshwater flows

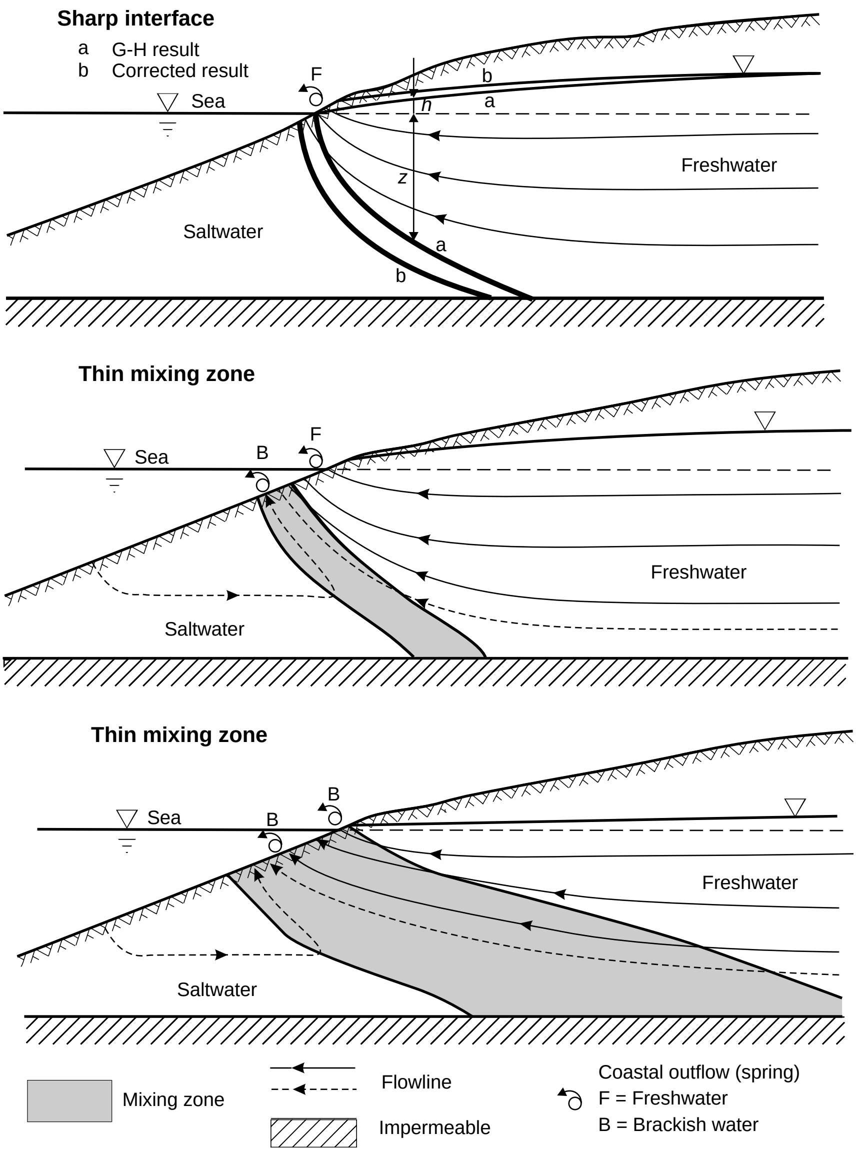

Figure 2. Freshwater–saltwater relationships in an idealized, homogeneous coastal water table aquifer in which main recharge comes from other areas to the right hand side. The vertical scale is exaggerated respect the horizontal one; the part above sea level is exaggerated respect the part below sea level, which explains the smallest apparent slope of continental surface. Actually $z = \alpha h$ in the upper figure, with $\alpha = 40$. The upper figure shows the sharp interface case: no mixing zone and steady saltwater. 'a' is for $z = \alpha h$, being h the water table elevation and z the depth of the interface; 'b' is for true heads along the interface (see Figure 1). The two other figures show the development of a steady state mixing zone. Inside the mixing zone the isoconcentration lines are more or less parallel to the zone boundaries and the flow lines tend to be also parallel to them. Brackish water is discharged along the coast, even above sea level for very flat water tables, which means that a small fraction of seawater must flow landwards.

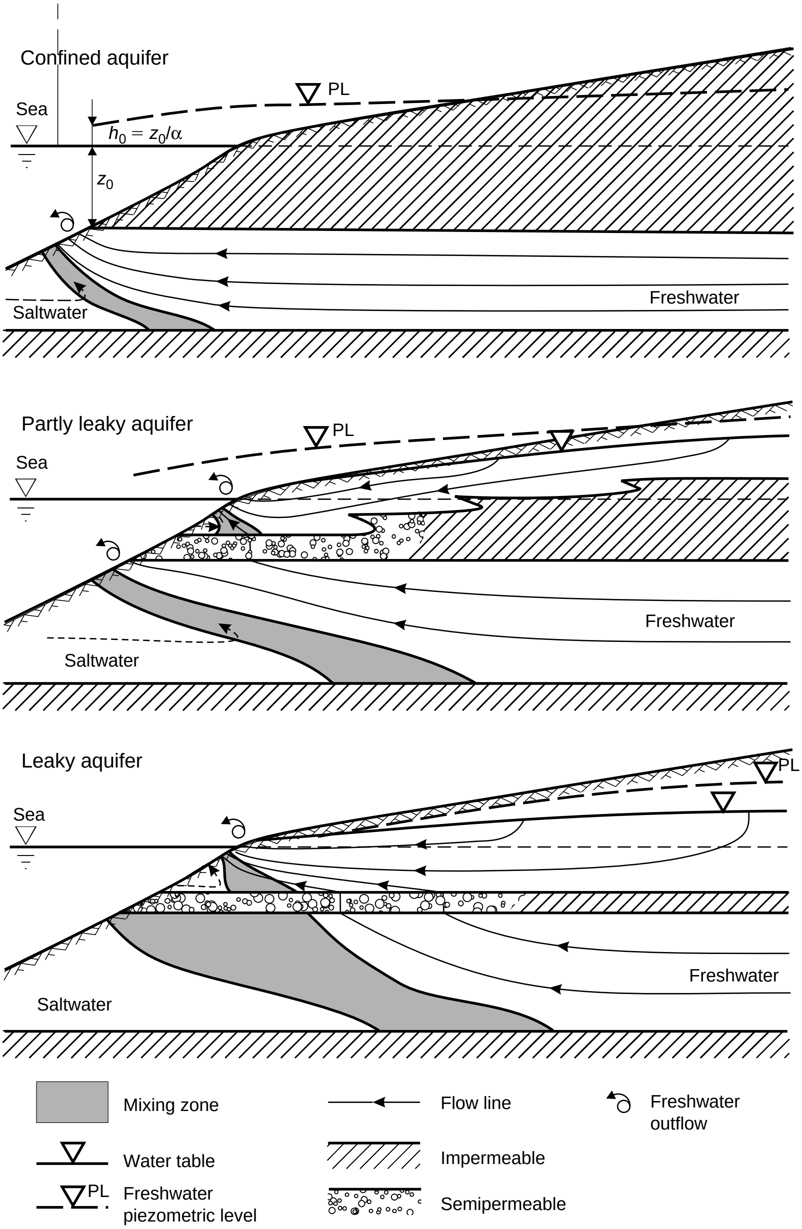

Figure 3. Freshwater–saltwater relationships in coastal confined aquifers. In the submarine outflow face the piezometric head of freshwater must overcome the denser seawater column; otherwise the discharge is not possible and the sea invades partly or wholly the confined aquifer. The upper figure corresponds to a fully confined aquifer discharging into the sea. The central figure shows the effect of a low permeability top layer that becomes thin near the coast, allowing upward discharge of freshwater to a locally recharged water table aquifer. In the lower figure the low permeability layer is thin, allowing confined flow to discharge to the water table aquifer, and there is no discharge to the sea. See caption of Figure 2 for comments on vertical scales.

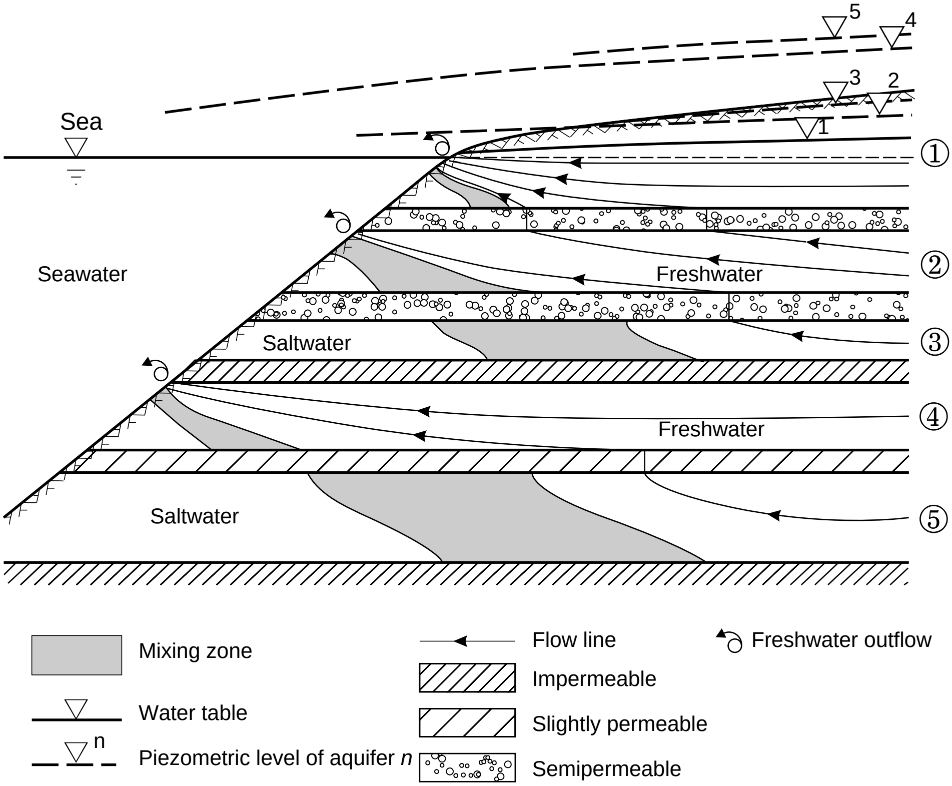

Figure 4. Freshwater–saltwater relationships in a thick multilayered coastal aquifer system. Aquifers alternate with variable low permeability layers. Each aquifer show a different saltwater and mixed water penetration, that partly depends on landward freshwater potential. A borehole will penetrate water bodies of different salinities, that vary accordingly to distance to the shore. See caption of Figure 2 for comments on vertical scales.

through the aquifer) and the other with a low upstream freshwater head (almost stagnant sea water in the aquifer, but with a small flow towards the mixing zone).

The assumption that there is steady sea water level and salinity distribution inside the aquifer system is true for short residence time formations. Otherwise the system may not be under steady conditions of water potential and especially of salinity distribution. It is well known that about 10–11,000 years ago the sea level was about 100 m lower than at present and has remained more or less stable at around the present level for only the last 6,000 years. This means driving forces and salinity changes in the ground that may still be ongoing in large and thick aquifers or in low permeability formations. Coastal changes due to sedimentation-erosion have a similar influence, as happens in quickly growing deltaic or subsiding coastal zones, or to the long-term changes in aquifer recharge due to climatic evolution and natural or man-induced land cover modification, (Lambrakis & Kalergis, 2001). The current trend for mean sea level is to go up, with forecasts of about 30–50 cm in the coming half-century. Problems of changing conditions of land elevation, both natural and man-induced, are well studied in the Netherlands (Kooi & de Vries, 1998; Kooi & Groen, 2000) where complex groundwater salinity problems exist.

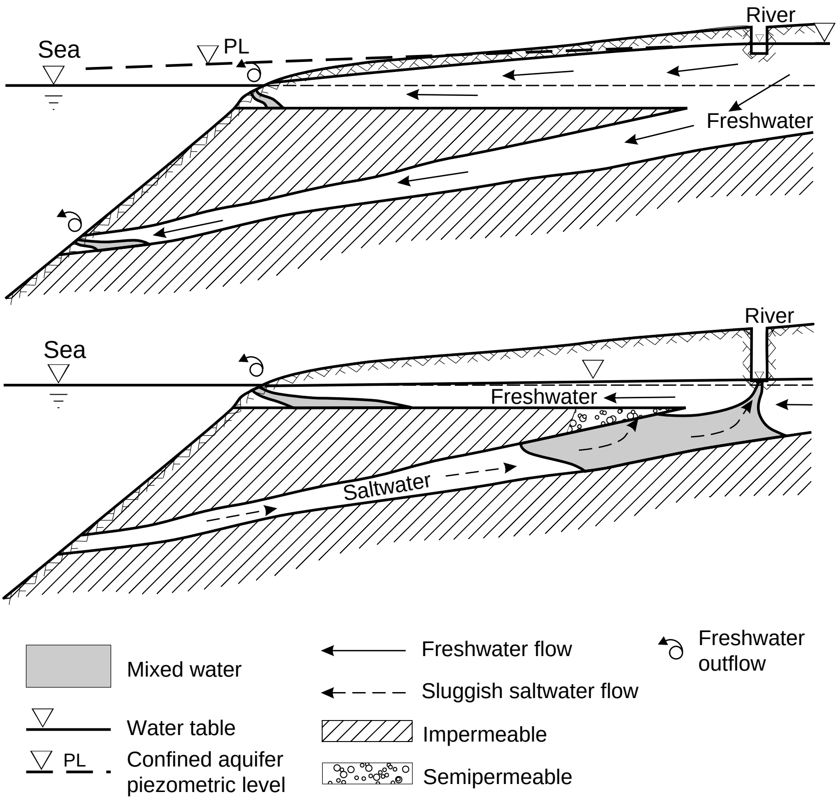

Figure 5. Schematic representation of a deltaic aquifer system with two aquifers, one phreatic and the other confined, which merge landwards, where the main river channel sets up the freshwater head. The discussion refers to the confined aquifer, considered here the most important hydraulic structure. In the case of the upper figure, inspired in the Lower Llobregat river valley and delta (Barcelona, Spain), the river channel altitude at the confined aquifer head is high and allows the flow of freshwater towards the sea outcrop and its discharge. The deep aquifer was seawater intruded early in the Holocene, when seawater rose to present position and invaded the estuary without the impermeable formation covering it. It is now flushed out with freshwater. The lower figure is inspired in the Lower Ebre river valley and delta (Southern Catalonia, Spain), in which the river channel is deeply incised by an important stream at the unconfined aquifer upper part. This means that there is no freshwater head for freshwater circulation, the aquifer contains early Holocene seawater. Besides seawater slowly flows upstream to mix with freshwater at the upstream part, where brackish water is discharged and flushed out by the main river and tributaries.

In any case the flow of freshwater exercises the important role of keeping the salinity distribution in the ground and of transporting advected and diffused salinity to the sea. From this viewpoint freshwater flowing out to the sea is not wasted if the goal is to maintain the current salinity pattern, to maintain water-dependent coastal wetlands and their salinity, and also to preserve some coastal sub-marine habitats that depend on the mixing with outflowing freshwater.

Coastal aquifers on relatively large islands do not differ essentially from continental ones except in the limitations imposed by the lack of large river basins and extensive recharge hinterlands. It can been considered that a small island has a surface area of less

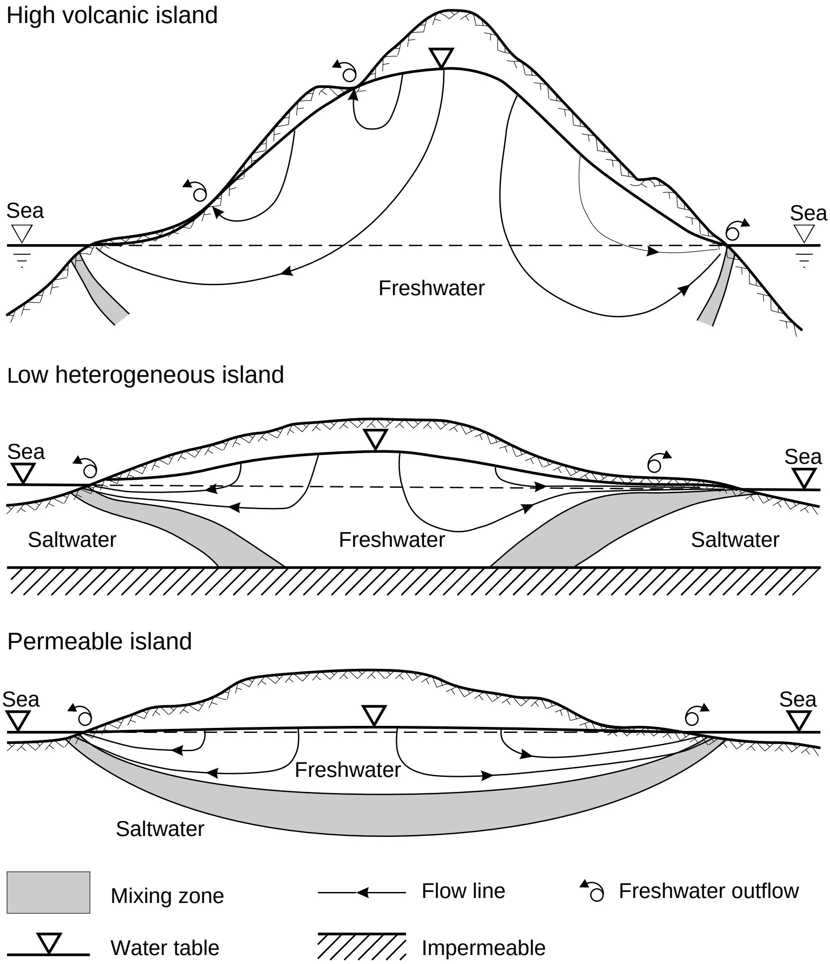

Figure 6. Small islands and capes, rounded or elongated, recharged by local rainfall. The upper figure, inspired in Gran Canaria volcanic island, shows a low permeability high island (internal structure may be quite complex) in which seawater intrusion is limited to a narrow coastal strip or coastal plains in recent volcanics, alluvium or slope deposits. The central figure, inspired in Mallorca island central corridor, corresponds to heterogeneous carbonate and marl formations, in blocks, in which there is a central body of freswater and deeply penetrating saline and brackish water wedges in the more permeable deposits. The lower figure, inspired in Malta island, show a floating freshwater lens situation in the lower highly permeable carbonates; the cover are low permeability chalky (globigerina) limestones.

than 2,000 km, although the shape plays also an important role in the distribution of saline water bodies. The main characteristics have been considered in detail in Falkland & Custodio (1991) and in the numerous references given in this book. There is a wide range of situations from cases in which the effect of direct sea water can be considered small in most of an island's aquifers, as in the volcanic Canaries Archipelago, to others in which salinity seriously affects a large part of the aquifers, as in Mallorca (Manzano et al., 2000), which is somewhat larger than a small island, or to the extreme case in which

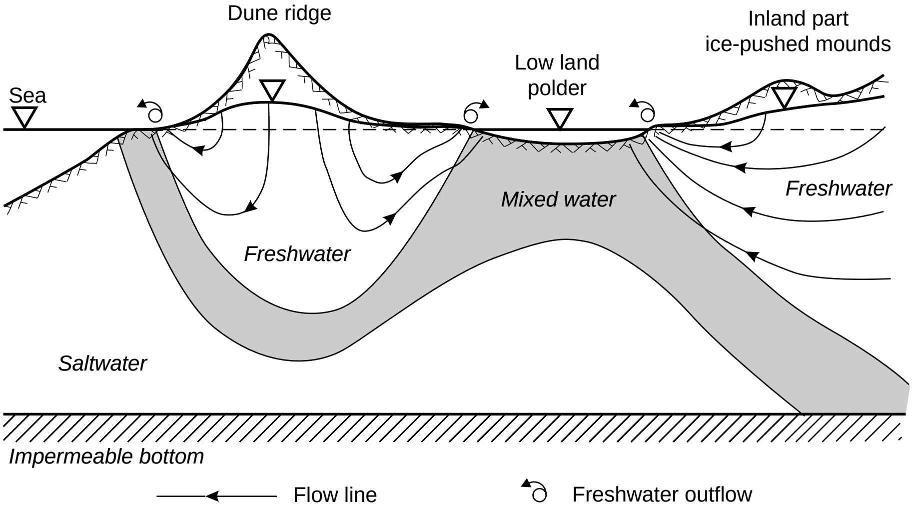

Figure 7. Situation in a coastal plain with a near-shore dune belt and intermediate low lands, as in the case of The Netherlands and Belgium, and also a series of coastal aquifers north and south of Mar del Plata. The existence of low permeability interlayers often play a key role in salinity distribution and the flow of brackish water to the low lands. The low lands may be evaporation ponds, or externally fed freshwater lakes, or in the case of The Netherlands artificially drained areas from which brackish water is discharged at low sea tide or pumped out; in recently 'reclaimed' areas current situation is unsteady and aggravated by peat layer subsidence.

there is a partly or wholly continuous body of sea water below the island. In this last case the body of freshwater is a lens floating on saline water, as in Malta and also in oceanic atolls (see Oberdorfer et al., 1970; Underwood, 1992 for additional information), key islands ('cayos') in the Caribbean sea and elongated sand bars along many coasts (as in eastern United States) and deltaic formations (see Figure 6). A floating body of freshwater may also be found in areas of continents and large islands, such as capes formed by very permeable formations, as in some areas of Florida.

The comments and discussions that follow refer to situations in which present sea level and marine water play the dominant role. But there are coastal and island aquifers in which salinity problems are mostly related to pre-Holocene marine water, as in many areas of Belgium and the Netherlands, or to present sea water, which may penetrate through top aquitards to slowly replace freshwater established in late Pleistocene times, when the sea level was about 100 m lower than at the present time and the sea coast was far away seawards, as is the case of Surinam (Groen et al., 2000) and other parts of Western Europe or the north-east corner of South America (Edmunds & Milne, 2001). Figure 7 refers to a coastal aquifer with a dune belt and lowlands behind. Figure 8 depicts the case of confined coastal aquifers that have no downflow discharge face directly into the sea bottom.

Ancient marine water, often concentrated by evaporation and suffering partial precipitation of salts, and later salt dissolution and dilution by continental freshwater, may be one of the sources of salinity of some coastal areas. This happens in Israel (Sharit & Furman, 2001; Vengosh & Ben-Zvi, 1994), where the formation of saline waters and brines are attributed to the Messinian (Miocene) crisis of the Mediterranean. In other cases diluted ancient marine water is trapped in highly heterogeneous formations that are

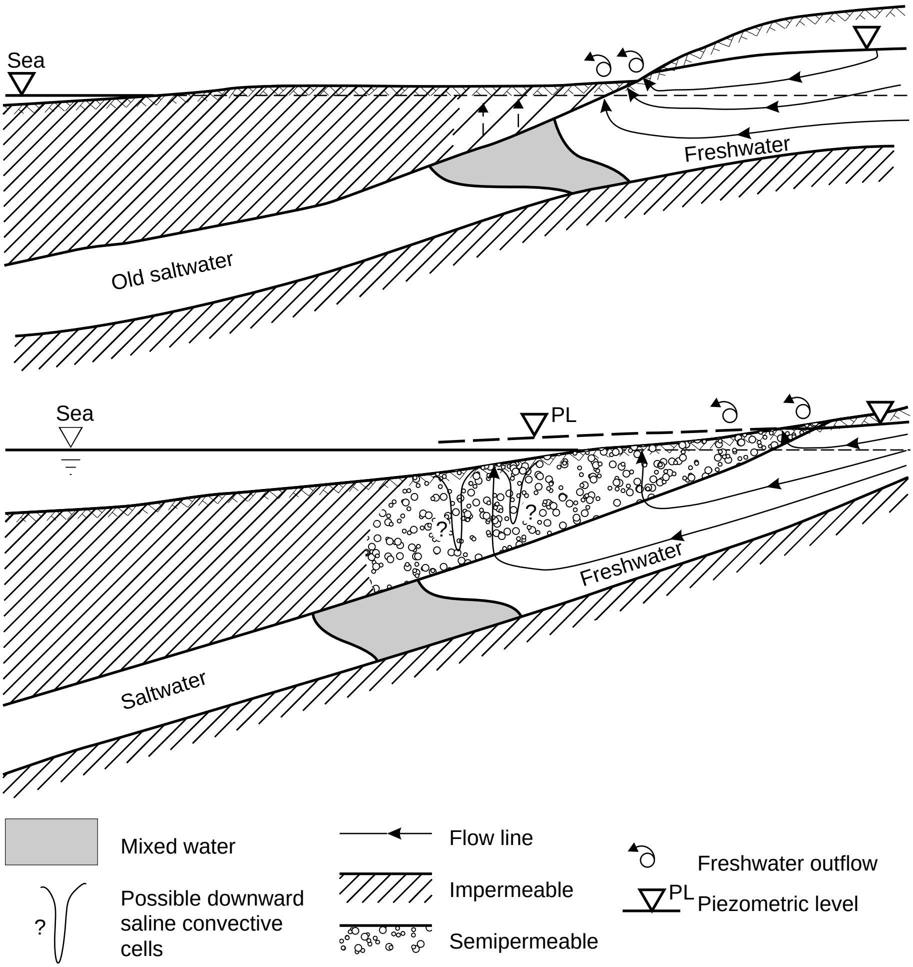

Figure 8. Two coastal confined aquifer situations. The upper figure is the case of a closed one in which recharge in the unconfined upstream part discharges inland; the rest of the aquifer may contain palaeowater, that may be fresh (as in the case of Aveiro, Portugal) or saline (as in the case of southeastern England) depending on past circumstances and the possible existence of permeable features in the impermeable confining layer. The lower figure shows a similar situation but with a semipermeable confining unit (inspired in northeastern Sudamerican situations) which in the late Pleistocene allowed freshwater penetration to displace preexisting water far offshore; with present sea level elevation this freshwater is a slowly renovating palaeowater or is being penetrated by convective vertical plumes of seawater in the offshore part if freshwater head in the aquifer is not enough to compensate for the unstable vertical salinity distribution.

regionally almost impermeable but with local permeable features that allow temporal discharges from wells, as in Fuerteventura Island, in the Canaries (Herrera & Custodio, 2002), or the high salinity is of climatic origin, as in many southern areas of the Canary Islands (Manzano et al., 2001).

Return irrigation water from crop cultivation plots in coastal areas may be quite saline if water use is highly efficient and may attain a total dissolved salt content of 2–5 g/l and even higher, as in south-east Spain and the Canaries; they may salinize groundwater resources that exist below the irrigated fields.

FRESHWATER DEVELOPMENT IN COASTAL AQUIFERS

Abstraction of groundwater from a coastal aquifer produces a groundwater head drawdown, both in the freshwater and the saltwater body. The immediate result is the creation of head depressions and the consequent reduction of freshwater discharge into coastal wetlands and riparian areas, and especially to the sea. This means an increased inflow of sea water and the modification of isocone (isoconcentration) contour surfaces. Saline water tends to approach the abstraction sites and there is a trend to slowly increase the thickness of the mixing zone at the time it moves (Figure 9). A new steady mean salinity distribution can be attained after some time. This period may be months to centuries long, depending on the aquifer characteristics and the location of abstractions. In any case an enlarged volume of the aquifer system will be occupied by saline and brackish water. Abstracted water will continue to be fresh or may become brackish and saline; the rate depends on the general behaviour of the aquifer and local circumstances. Lateral (mostly horizontal) displacement of saline water is a slow process while upward vertical movement (upconing) may be very fast (hours to days) in the absence of low-permeability horizontal layers. The presence of low-permeability layers may play a key role in delaying and reducing saline water upconing (Motz, 1992).

A three-dimensional description is often needed to correctly describe what happens, taking into account the vertical distribution of heads and salinity. An adequate monitoring network is needed, and this will be more complex and expensive than in common continental aquifers.

It is possible to abstract freshwater from a coastal aquifer that has saline water below if well discharge and penetration are small. Otherwise there may be brackish and saline water upconing (Figure 10) that may degrade pumped water or it render unusable.

Actual situations can be found in which there is intensive exploitation of a coastal aquifer with wells abstracting brackish water alongside those pumping freshwater. The situation may be quite complex and depend on local circumstances (Figure 11). Sea water encroachment is not homogeneous but follows permeability features and the spatial exploitation pattern (Figure 12), and varies with sea water inflow rates (Figure 13).

COASTAL AQUIFER MANAGEMENT

The purpose of coastal aquifer management is the same as for other aquifer systems – to achieve a sustainable use of groundwater, coordinated with the use of other water resources, to meet part of the demand for water by supplying water of adequate quality, in the place at the right time, respecting environmental and habitat restrictions. The main additional items to be considered are the risk of salinization and water quality degradation in relation with the possible accumulation of manmade contaminants in areas of low hydraulic gradient and flow pattern forming a closed area due to groundwater abstraction conditions. Often these risks do not result in immediate threats, but the results may be delayed for a long time. This means that coastal aquifer management should rely on conservation and protection measures. This is not new for aquifer management, but for

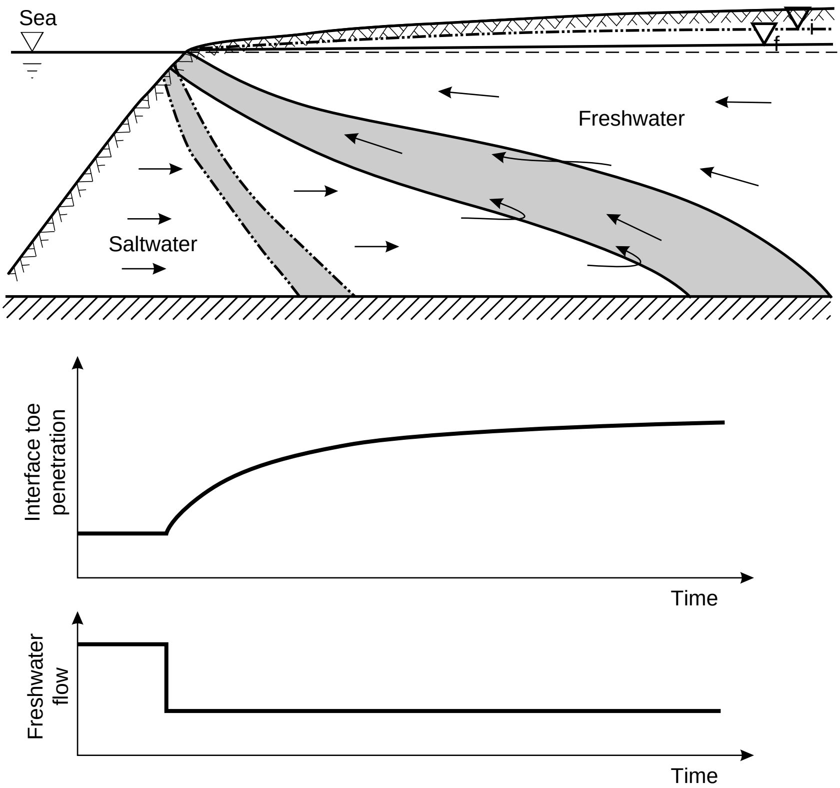

Figure 9. Effect of a reduction of freshwater flow at the coastal area. This means that the seawater wedge moves landwards to a new position at the time the mixing zone grows. The coastal freshwater storage decreases as well as the risk that near-the-shore wells may be affected by saline or brackish water. During the transient period the former freshwater part of the aquifer is invaded by saline water. This means cation exchange processes that for a given mixing degree makes harder the resulting water (especially in Ca) and depleted in alkaline ions (for Na the effect may be relatively small due to its high concentration, but it is more clear for K, so the ratio Na/K increases). The Ca increase may induce calcite precipitation and if the increase is very high (in high salinity waters in high cation exchange formations) it may happen that some gypsum precipitates, reducing the ratio SO_4/Cl. This gypsum may dissolve afterwards. The behaviour of calcite is more complex since activity coefficients behave non-linearly in the mixing. At medium to high salinities calcite, and especially aragonite, may dissolve.

coastal aquifers monitoring and protection measures should be more strictly and carefully planned. Management means a compromise between meeting the water demand and limiting the demand without serious damage to the local economy and social needs.

The main requirements for sound coastal aquifer management are:

- good understanding of aquifer behaviour
- adequate monitoring systems, with early warning signals and public information
- the authority and popular will to attain sustainable use
- a water management institution with adequate tools and resources
- the effective participation of stakeholders in management
- education, training and dissemination of knowledge and data.

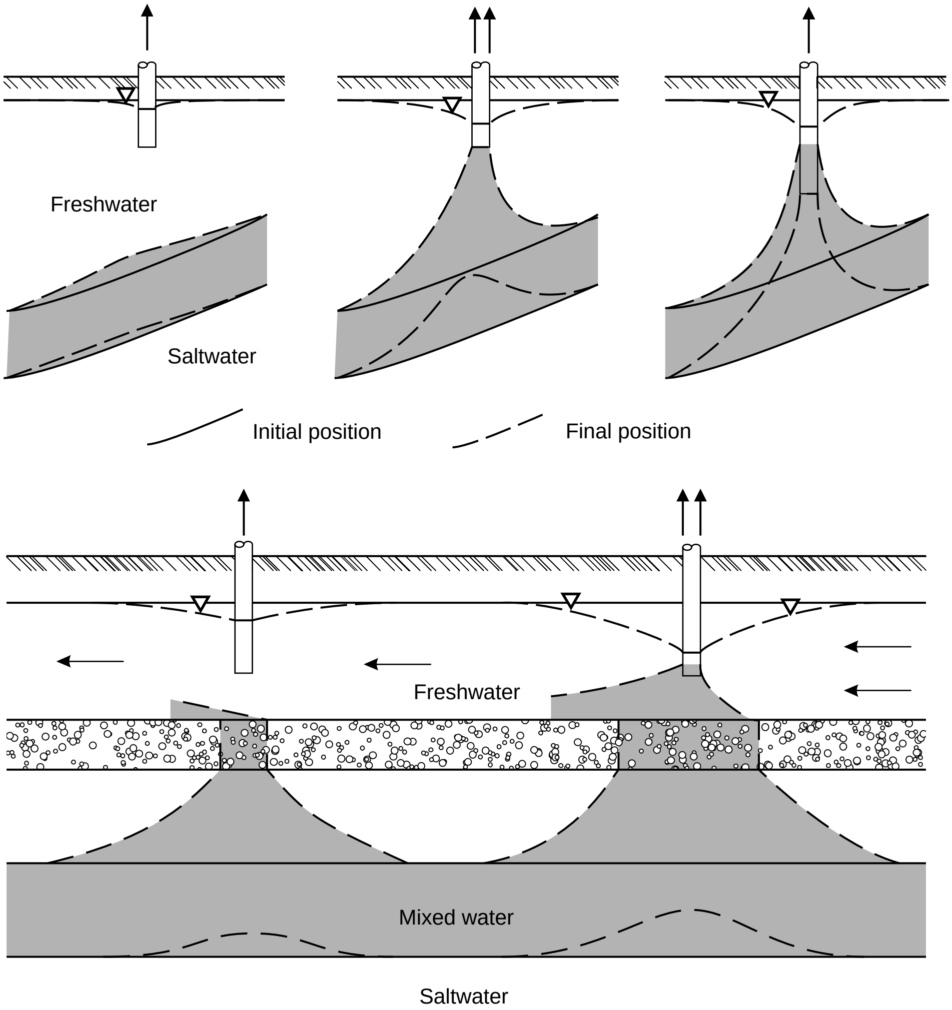

Figure 10. Abstraction of freshwater with brackish and saline water below. There is upconing of high salinity water toward the well or drain (or lake, reclaimed land, polder or drained excavation). This may be a relatively fast process. The well penetration and discharge rate control the result. The presence of low permeability interlayers spreads the influence and delays the process.

There are scientific and technical issues, but the decisive ones are economic, social, administrative and legal in nature. Quantitative approaches are possible after a good conceptual model is available, which generally needs, besides hydraulic data, hydrogeochemical and environmental isotope studies. Water samples should be obtained not only from existing wells and boreholes but from short-screened boreholes especially constructed to monitor the brackish and saline water bodies and the aquitards.

The tools that are currently available allow numerical modelling under common situations, even in three-dimensions if that is really needed. Numerical modelling is greatly reinforced by considering mass transport (salinity and individual components)

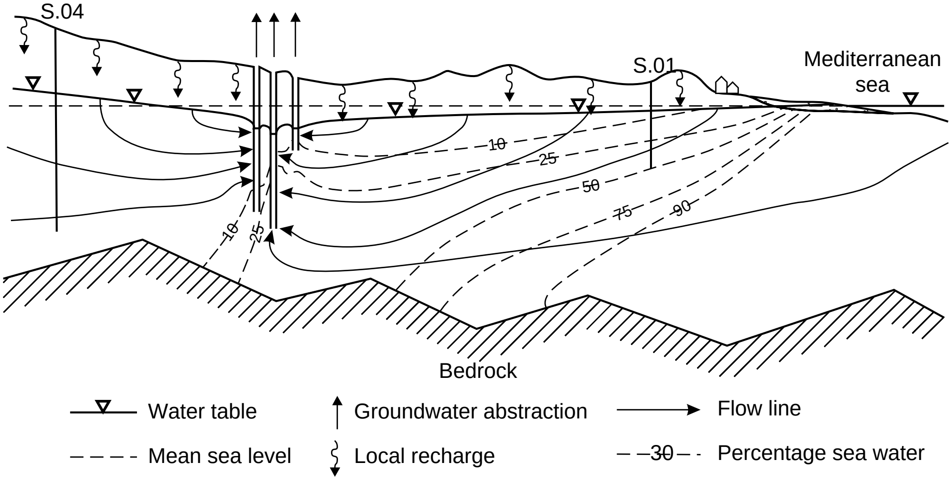

Figure 11. Sketch of freshwater–saltwater mixing in a coastal aquifer of high permeability which is intensively exploited in inland areas. Abstracted water is in part freshwater from the continental side and partly marine water arriving together with recharge in the coastal strip (after Pascual & Custodio, 1990). The mixing in the wells is variable according to location, well depth and exploitation regime.

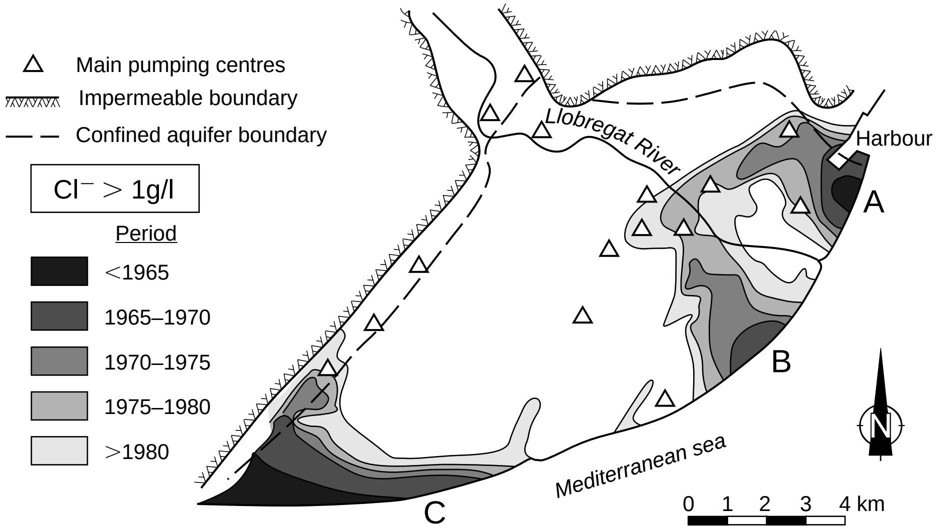

Figure 12. Progress of sea water encroachment in the confined aquifer of the Llobregat delta, Barcelona, as a consequence of abstractions (modified after Iribar & Custodio, 1993). The areas correspond to a chloride content higher than 1 g/l at different times. Three main encroachment fronts exist: A – corresponds to coarse piedmont and alluvial fan sediments; B – corresponds to the late Pleistocene river channel; C – is an area of low altitude, recent sediments, in which marine water was being displaced; the process is now reversed as a consequence of groundwater exploitation and there is saline water encroachment through the more permeable materials near the delta boundary. These areas are separated by less permeable sediments. Encroachment is delayed in them. The confined aquifer outcrops about 4 km offshore, at about 100 m below present sea level.

Figure 13. Evolution of seawater encroachment through the submarine outcrop of the confined aquifer of the Llobregat delta, Barcelona, NE Spain (see Figure 12), after Iribar et al. (1993). It is obtained by automatic calibration of a 2-D homogeneous fluid numerical simulation of flow and salinity transport, using 20 years of historical data (1965–1986). Groundwater abstraction increased until 1975 and later on decreased, especially in the upstream and eastern part. The changes are quite fast since it is a small (80 km^2), highly transmissive confined aquifer that is intensively exploited. The maximum rate of seawater encroachment was 30 Mm^3/year, that was reduced to 8 Mm^3/year in 1985, or about 10% of total groundwater abstraction. It concentrates in some highly saline water wells used for industrial cooling. This explains that the system tends towards a steady state when the abstraction pattern does not change and the recharge sources are protected.

besides groundwater flow transport (see Bobba, 1993; Sandford & Konikow, 1985; Voss, 1985; Voss & Souza, 1987; Kipp, 1987; Molson & Frind, 1994 for further details on basic principles). Simmons et al. (1999) explains the application to the complex situation of the Netherlands.

The change in salinity produced by the intrusion of sea water and other saline waters, when it can be monitored, gives clear signals on what is happening through conspicuous processes of ion exchange and the related dissolution-precipitation of minerals (Beekman, 1991). Some of these appear with small changes and thus may be used as early warning signals (Van Dam, 1997).

Coastal aquifer system management has to take into account the following considerations.

- Fresh groundwater abstraction means reducing freshwater discharge into the sea or estuaries, with a parallel increase in the volume of the groundwater system, which is filled with brackish and saline water, and some redistribution of groundwater bodies.
- Under given restrictions, an optimal quantity and pattern of freshwater abstraction can be found that is a trade-off between the benefits from freshwater availability and the cost of reducing freshwater storage, affecting habitats and abandoning some of the existing wells and water infrastructures. The result is not an exact figure but a series of possibilities, that also may change with time.

- Collective management is needed. This means that individual stakeholders' water rights have to be totally or partly given to a water management institution or users' association, which proposes and enforces water shares under some equity rules (Hernández-Mora & Llamas, 2001).

The number of wells that may be affected by salinization for a given coastal aquifer system development depend not only on total aquifer abstraction but also on the circumstances of each well, such as discharge, penetration, distance to the coast, abstraction regime and local aquifer characteristics. Upconing of saline water – be it recent or old marine water, or deep-seated saline water – is a common process for deep wells and intensively exploited wells. Sometimes it is not easily identified.

Often trends that are easy to observe are sought to define when there is excessive groundwater abstraction. It is often called overexploitation, although this term is not a useful one and should be abandoned (Custodio, 2002). There are trends that are not easy to observe – or plain facts – that can be substituted for a sound aquifer system study. Such a study is often justified when the benefits derived from groundwater exploitation and sustainable use have to be considered and a permanent monitoring network to update the information is needed. Intensive development of aquifers produces social benefits as well as other issues and costs that need careful study if valuable opportunities are not to be lost (Llamas & Custodio, 2003), and this is especially true for coastal aquifers.

A continuous drawdown over some years is not a sure sign of pumpage exceeding recharge but may be the result of the transient stages between the initial situation and a new one. The application to a coastal aquifer is more complex since head changes may not be clearly seen in practical terms due to the positive hydraulic barrier effect of the sea, but groundwater quality degradation due to the admixture of saline water may occur. Any freshwater abstraction from a coastal aquifer means a landward and upward displacement of saline water, as well as an expansion of the mixing zone. Then, some wells may be affected by salinization well before total abstraction approaches recharge in average terms.

Poor coastal aquifer management can be linked to the following factors.

- Too much abstraction relative to recharge, even if recharge is not exceeded. This happens more often when high-yielding wells can be drilled, especially close to the shore. Abstraction capacity is easily mistaken by non-experts as aquifer freshwater resources. Strict control of well use is needed.
- Wells and drains are too close to the shore. Salinization is often the consequence of saline upconing. Placing the wells far from the shore helps to prevent sea water contamination, but exploitation costs are higher and longer distribution mains are needed when the water demand centres are close to the coast.
- Individual wells pumping too much water. To avoid upconing and/or lateral saltwater encroachment discharge has to be limited, sometimes to a small fraction of the well yield. This is seldom understood by abstractors, but has to be enforced by management rules and supported by a groundwater users' association.
- Poor well construction or design, such as excessive depth, screens facing salinized or easily salinizable layers, and poor protection against corrosion around saline water aquifers or aquitards. It may be necessary to close down or partially grout some wells.

Large confined coastal aquifers may still contain freshwater recharged during the Pleistocene low sea level stand (palaeowater) and this water may be an important reserve for human supply if correctly exploited and managed (Custodio et al., 2001).

COASTAL AQUIFERS AS COASTAL HYDRAULIC INFRASTRUCTURES

In many cases coastal aquifers have a large water storage capacity. This capacity is mostly in unconfined aquifers but can also occur in deeper formations with saline water if it can be effectively replaced by freshwater. Freshening a saline water aquifer may be a complex problem. Aquifer exploitation may increase the storage capacity of the unconfined aquifers by depleting the water table in areas not controlled by a nearby base level.

Therefore coastal aquifers may become important elements for human water supply and guarantee that if they are properly managed. Surplus water that is available at a given moment may be stored to be used later or freshwater storage may be depleted in periods of high water demand or in emergency situations, to be restored naturally or artificially during periods of low water demand and/or increased recharge. An important aspect is the ability to use freshwater storage in the ground for emergency situations. The main difference to continental aquifers is the need to monitor and forecast the saline water position and its slow movement in the aquifer system, and distribute wells and groundwater abstractions in such a way that the risk of saline water contamination is minimized. This means not only adequate monitoring and the operation of simulation models by a responsible institution, but also the capacity to decide when and where to abstract and recharge water in the whole aquifer system. Moreover, the part of the aquifer with saline water, or temporary intruded sea water, will need an excess of freshwater – often a large excess – to flush out trapped saline water in heterogenities and to restore sorbed ions on exchange sites.

Artificial recharge is one of the means to operate coastal aquifers as hydraulic infrastructures. There is enough experience to solve most practical problems related to recharge water, the methods to introduce water into the ground, the maintenance of recharge capacity and the recovery of introduced water (Custodio, 1986). The design and operation is something that depends largely on local conditions and must be tailored to each particular situation.

The effect of artificial recharge and the results of management activities can be numerically modelled provided adequate monitoring exists, but the operation of the facilities needs local experience and sometimes have to been preceded by pilot projects and plants. This also happens during the design, construction and operation of many other hydraulic infrastructures, for which the time and economic resources invested are considered part of the project and carried out carefully. The main difference is that the aquifer already exists and does not have to be constructed.

Often people take natural infrastructures for granted and consequently do not consider they have a value. Economically this is not true. The value of an aquifer system that is capable of performing a series of beneficial tasks can be estimated as the cost of the cheapest alternative infrastructure to perform the same task. In many coastal areas, where land is expensive, there is little available space for the construction of hydraulic structures and aquifers become especially valuable elements that also have a strategic worth.

The relatively small (80 km) but highly productive (up to 150 mm in 1975) lower Llobregat Aquifer (the lower valley and delta), with an associated regulation capacity of about 100 mm (Custodio, 1992), is an important element for supplying water for human and industrial use to a proportion of the four million inhabitants of Metropolitan Barcelona, in north-east Spain, and also provides key emergency storage of freshwater. Recharge and water abstraction facilities are threatened by the high pressure for urban and industrial expansion, and for new space for roads, railways, the airport and the harbour. The aquifer's value is reckoned at 300–600 million Euros, which justifies the effort to sustain its use and to force public authorities to invest in protection and rehabilitation, coordinating with the existing effective groundwater users' association.

Individual stakeholders does not generally have any idea of the value of the system they are using and are not aware that they share the aquifer with many others. Only the combination of institutional and collective participation in management can put things in real perspective and direct economic activity towards sustainable use. This is the means of avoiding the 'tragedy of common property' – one of the 'hydromyths' to be ousted (Custodio, 2002).

Strategies for coastal aquifer exploitation may include using brackish or saline water, either intentionally or as the result of other freshwater sources being unavailable. Some benefits are derived from using these brackish or saline waters. They can provide water of almost constant temperature, advantageous for cooling – after coping with the greater corrosiveness and hardness – or be the source of water for desalination. This groundwater is already filtered, has no fouling micro-organisms and has relatively stable physical and chemical characteristics. After correcting for the presence of possible undesirable contents, like high hardness and high silica content, this may be a cheaper alternative to using sea water directly, especially if groundwater is brackish and no large salinity changes are expected during exploitation. In this case pressure membrane processes lower than for sea water are employed for desalination and especially reverse osmosis.

Groundwater with a salinity close to that of the sea water may be used to supply fish farms. But in many circumstances such waters are anoxic, with some ammonia (as in deep deltaic aquifers rich with organic matter) and other deleterious ions, such as heavy metals and especially As in some cases. This limits their direct utilization and the benefit of getting a constant temperature source that saves energy may be reduced.

Pumping saline water from wells decreases saltwater head in the aquifer and consequently limits saltwater encroachment in the aquifer, even when freshwater heads are maintained below mean sea level. This prevents salinization of other landward wells, but is a temporary solution. The abstracted saline water may have a beneficial use, such as cooling for industrial plants. The use of brackish or saline groundwater, directly or after being mixed with freshwater to reduce salinity, presents a series of drawbacks:

- it jeopardizes the possible re-use of treated waste water when saline water is disposed into or penetrates the sewerage system
- it may decrease agricultural output and produce soil salinization and alkalinization
- it may impair the water-table aquifer quality, either through leaks in distribution mains and pipes, or in the disposal pipes and canals to the sea, or by generation of saline return irrigation flows.

Direct coastal aquifer protection measures, beyond improved management and imported new freshwater sources for the area, include barriers to control sea water encroachment. They are discussed elsewhere (Custodio & Llamas, 1983; Custodio & Bruggeman, 1987) and some details and also relevant references are given in Custodio (1986).

Physical barriers have been envisaged and projected repeatedly for small, well-bounded coastal alluvial formations. But it seems that no significant physical barrier has been constructed up to now. The cost is very high relative to the small increase in water resources such barriers provide, besides construction problems, doubts about efficiency and associated quality problems, due to impaired exportation of solutes.

Hydraulic barriers consisting on a line of injection wells have been constructed in California, the oldest of which is near Los Angeles (Bruington et al., 1987; Bruington & Seares, 1965). This is an expensive solution to sustain domestic supply wells in large areas of low-density housing, but the construction of a water supply network in extensively salinized areas was an even more costly solution. The barrier restores freshwater outflow to the sea, provides flushing of inland trapped saltwater and produces a net aquifer recharge. In the same area saltwater pumping barriers or combined freshwater injection and saltwater pumping barriers have been experimented with, the design being adapted to local aquifer circumstances. Since a large part of the injected freshwater is to restore the flow to the sea, carefully treated tertiary sewage effluent has been used in some cases, as near Los Angeles, California (Argo & Cline, 1985, in Asano, 1985) or is being planned in other areas, as Metropolitan Barcelona.

These are solutions adapted to local circumstances. They are not necessarily the right solution for other aquifers. For the confined aquifer of the Llobregat Delta, Barcelona, currently the best solution seems to be temporal pumping of saline water in the more permeable areas, restoration of recharge in the lower valley and substitution of some groundwater pumping stations by supplying freshwater from other sources (Custodio, 1992, 1993).

REMEDIAL ACTION FOR IMPAIRED COASTAL AQUIFERS

Coastal aquifer salinization, except for local upconing, is a slow process that involves the replacement of large volumes of freshwater by brackish and saline water. Remediation is therefore a slow process that includes not only replacing the volume of salinized groundwater for freshwater but also the need to flush out saline water trapped in heterogeneities and to replace the sorbed cations. This means high costs and long time, in contrast to the rapid results sought by investors and politicians, and expected by the media and lay people, and also by hydrogeologists who only consider homogenous groundwater flow. Again preservation appears as a necessary management goal. Well-documented experience on remediation is still scarce and mostly refers to small, highly permeable alluvial formations and karstic aquifers. Non-reactive displacement can be simulated (Travis and Song, 2001), but full consideration of ion exchange processes and changes in formation permeability are still ongoing.

Abandoning a coastal aquifer may be technically and economically a sound decision, if accompanied by the provision of a new freshwater source. But from the point of view of securing water supply, and freshwater storage for emergency situations, and of strategic value, aquifer abandonment is not generally a sound decision. In fact the recent European Union Water Framework Directive, ratified in 2000 and now being incorporated into national legislations, and developed into specific directives, sets out the need to restore and remediate aquifers. The particular application of this principle to already deteriorated coastal aquifers is still developing.

The abandonment of a former intensively exploited aquifer is followed by water table recovery and salinization of the previously drained vadose zone. This may mean significant damage to land use and urban infrastructures, as in the case of Mar del Plata, Argentina and Barcelona, Spain (Bocanegra & Custodio, 1995).

Remediation involves a series of actions such as careful grouting and isolation of wells that allow the movement of saline water from one layer to another, closing of pumping wells, enhancement of recharge, artificial recharge, forced drainage of saline water and the elimination of saline water disposal and leakage. This should be accompanied by an adequate monitoring network.

CONCLUSIONS

Coastal aquifers can be a sustainable source of freshwater if correctly managed and exploited according to recharge and local hydrogeological characteristics. Freshwater outflow to the sea controls sea water wedge penetration and the thickness of the mixing zone thickness. Management strategy has to decide how much freshwater is left for this role and to sustain groundwater-related habitats, and adapt the abstraction pattern to the resulting saltwater wedge penetration. Transient periods can be long and produce the false feeling that a given exploitation pattern is safe. As part of the long-term water planning of a region, freshwater in a coastal aquifer can be intentionally mined for beneficial use for a limited time, after which the aquifer becomes salinized; then other sources of water have to be brought in or water use has to be reduced (Llamas & Custodio, 2002).

An adequate monitoring network is needed to measure and take samples of fresh, brackish and saline water. A set of point boreholes for each site is the correct solution when vertical head gradients are to be expected or created by groundwater exploitation. Long screened boreholes may easily disturb salinity stratification.

Coastal aquifer management is not only a technical issue but an administrative, social and legal issue as well, in which aquifer water users have an important role once common goals are settled and individual rights are adapted to them. The local legal framework and administrative regulations condition how things can be solved. But in any case the right solution for management and sustainable use seems to be power sharing between a public institution and stakeholders with collective participation.

Aquifer protection implies protection of the role of low permeability layers, adequate well and borehole design and construction, and the plugging and sealing of abandoned wells and boreholes. Protection does not mean that every well or water right has to be protected. Management implies a trade-off between allowable saltwater penetration and sustainable groundwater abstraction in a dynamic framework.

Management and protection issues when the saltwater source is the sea are different from situations in which there are other sources of salinity. Identification of the true source of salinity and understanding of salinization dynamics is key starting points for management decisions, protection and remedial measures.

Remedial measures include the restoration of enough freshwater discharge to the sea and allowing enough time for saltwater displacement and diffused salt in low permeability heterogeneities to be flushed out, and for ionic exchange complexes to equilibrate again. Saltwater replacement can be forced artificially by saline water pumping and disposal into the sea, but at a cost. When human activities have changed the aquifer by actions like drilling boreholes through low permeability layers, constructing multiple screened wells or over-excavating canals and trenches, remedial action has to consider the possibility of restoring the natural situation.

Coastal aquifers are not only permanent sources of freshwater but storage reservoirs in areas where other storage reservoirs are often difficult or costly to construct. This is important for emergency purposes (droughts, breakdowns or pollution of other water supply sources) and has a strategic value. Coastal aquifers can be over-pumped for some time without undesirable results due to the slow movement of groundwater, provided there is no saltwater below the wells or there are natural clay layers that delay saltwater upconing.

Saline water in the ground may not easily be displaced by freshwater. Freshening may be a slow process that needs several equivalent volumes of freshwater, depending on salt diffusivity from low permeability heterogeneities and the total cation exchange capacity.

Physical, chemical, demographic, economic, geographical, social, legal and political circumstances make every coastal aquifer unique. Managerial solutions to its problems are also specific. This means that solutions have to be tailored to the aquifer characteristics, using all the knowledge and experience derived from other areas, but not simply by copying them. Otherwise expensive errors that are difficult to correct can be made. The most important step is obtaining a representative and workable conceptual model of the coastal or island aquifer system flow and salinity transport. Development, preservation and restoration activities must fit within this.

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Section 2

Workshops

CHAPTER 4

Strengthening the use of isotope techniques for a sustainable groundwater management

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ABSTRACT: Sustainability of water resources and vulnerability of groundwater to contamination have become important environmental issues in many countries. Only recently, water managers have realized the significance of appropriate evaluation of aquifer systems in order to improve the sustainable management of such systems and to avert deterioration of water quality. Isotope techniques, an independent and powerful tool, are becoming an integral part of many hydrological investigations and sometimes a unique tool in groundwater studies. The International Atomic Energy Agency (IAEA) fosters the role of nuclear science and technology to support sustainable human development. An overview of the groundwater sustainability problems worldwide, application isotope techniques for water resources development and management as well as the role the IAEA in the water sector, with an emphasis on Latin American region, are discussed briefly.

INTRODUCTION

Steady increase in global demand for fresh water, coupled with rapid industrial and agricultural development, is threatening the quality of fresh water supplies, especially in developing countries. Sustainability of water resources and vulnerability of groundwater to contamination have become important environmental issues in many countries. The availability of proper hydrological and hydrochemical information, before decisions are taken, is expected to lead to an improved management of the resources. Nuclear and isotope methodologies in hydrological studies provide powerful tools to hydrologists and civil engineers involved in water resources assessment and management.

The isotope hydrology section of the International Atomic Energy Agency (IAEA) has played a key role in the development of isotope hydrology, covering both theoretical aspects and the use of proven isotope and nuclear techniques to practical hydrological problems.

GROUNDWATER SUSTAINABILITY

In the Millennium Declaration, the UN member states resolved 'to halve by the year 2015 the proportion of people who are unable to reach, or to afford, safe drinking water' and ‘to stop the unsustainable exploitation of water resources, by developing water management strategies at the regional, national and local levels, which promote both equitable access and adequate supplies’. Some forecasts show that, by 2025, more than 3 billion people will face water scarcity. But this is not because the world lacks water. As HRH the Prince of Orange said to the panel of UN Secretary General in preparation for the Johannesburg Summit, the world water crisis is a crisis of governance. At the global scale, there is probably enough water to provide water security for all, but only if we change the way we manage and develop it. In South America, for example, water resources approximate 3 millions km^3 and only the equivalent of 1/10 of the total amount of water contributed by precipitation is used every year (GWP-SAMTAC, 1999). The major problems these countries face are sustainable groundwater use and prevention of contamination of the available resources.

Increase of water shortages and quality deterioration has a great impact on economical and social development of the countries. Groundwater represents about 97 per cent of the fresh water resources available in the world, excluding the resources locked in polar ice (World Bank, 1999) and is the main source of drinking water in many countries. However, after some years of groundwater development, it has been observed that pollution levels show a steady increase. This is usually associated with an excessive abstraction also causing a decline in the water table. Despite the importance of groundwater for many societies, there is not enough public concern about its protection, perhaps because the extent and availability of groundwater are not easily measured. In some cases, it has been found that the exploited groundwater is not a renewable resource, thus leading to the mining of the resources. The impact of increasing degrees of temporal and special climatic variability on water resources is also an important consideration.

In the formulation of sustainable management strategies the following knowledge requirements arise: i) determination of the aquifer recharge rates and their temporal and spatial variation (especially in arid and semi-arid environments); ii) evaluating the age and origin of groundwater explored or abstracted and associated contaminants; iii) determination of groundwater flow-fields; iv) assessment of the spatial variations in aquifer vulnerability in relation to land use; and v) identification of the three-dimensional distribution of deep, high quality paleaeo-groundwater bodies, which represent potential strategic reserves.

Hydrogeology, and related scientific disciplines, provides a wide range of methods to address these questions. However, due to the complexity and inaccessibility of the subsurface, coordinated use of independent methods is required to arrive at a consistent and robust conceptual model of the physical and chemical characteristics of the groundwater system. In establishing such models, environmental tracers are extremely useful and, in some cases, the only means for obtaining the necessary knowledge. To guarantee that aquifer management profits from environmental tracer data, the interface of data information to management models has to be strengthened. A number of good examples show that environmental tracer information can contribute to water resources assessment, planning and management.

The following issues appear to be most important for sustainable groundwater management in South America (GWP-SAMTAC, 1999):

- lack of information in relation to hydrogeological and hydrochemical characteristics of aquifers and absence of monitoring these parameters

- intense urbanization without any regulation and insufficient infrastructure for water supply and wastewater network
- poor quality of cesspools and septic tank construction
- saline intrusion in coastal zones
- inadequate or non-existent management of industrial and mine waste deposits
- the erroneous idea that groundwater is a common resource that should be used freely and without any restrictions leading to over-exploitation of many aquifers
- absence of integrated management regulation or organization in the countries
- lack of numerical models for adequate evaluation of the resources.

In Central America, suffering from droughts and impacts of climate change, the issue of groundwater management is even more relevant.

ISOTOPES IN HYDROGEOLOGICAL INVESTIGATIONS

A comprehensive understanding of a hydrogeological system is necessary for a sustainable resource development without adverse effects on the environment. Isotope techniques are effective tools for fulfilling critical hydrologic information needs. The cost of such investigations is often relatively small in comparison to the cost of classical hydrological techniques, and in addition, isotopes provide information that sometimes could not be obtained by other techniques. Stable and radioactive environmental isotopes have now been used for more than four decades to study hydrological systems and have proved particularly useful for understanding groundwater systems (Aggarwal et al., in press).

Applications of isotopes in hydrology are based on the general concept of 'tracing', in which either intentionally introduced isotopes or naturally occurring (environmental) isotopes are employed. Environmental isotopes (either radioactive or stable) have the distinct advantage over injected (artificial) tracers in that they facilitate the study of various hydrological processes on a much larger temporal and spatial scale through their natural distribution in a hydrological system. Thus, environmental isotope methodologies are unique in regional studies of water resources to obtain time and space integrated characteristics of groundwater systems. The use of artificial tracers generally is effective for site-specific, local applications.

The most frequently used environmental isotopes include those of the water molecule, hydrogen (^{2}H or D, also called deuterium, and ^{3}H, also called tritium) and oxygen (^{18}O), as well as of carbon (^{13}C and ^{14}C, also called radiocarbon or carbon-14) occurring in water as constituents of dissolved inorganic and organic carbon compounds. ^{2}H, ^{13}C and ^{18}O are stable isotopes of the respective elements whereas ^{3}H and ^{14}C are radioactive isotopes. Table 1 presents a list of isotopes used in groundwater studies.

Variations in stable isotope ratios of natural compounds are governed by chemical reactions and phase changes due to the energy difference between chemical bonds involving different isotopes of an element. Such energy differences are caused by the relative mass difference between isotopes. The stable isotopes of light elements show greater variations because they have larger relative mass differences. Stable isotope ratios in hydrology are conventionally reported as per mil (‰) deviation from those of a standard using the δ (delta) notation. The isotopic standard used for hydrogen and oxygen isotopes is the Vienna Standard Mean Ocean Water (VSMOW). The IAEA distributes stable isotope reference materials to all interested users.

Table 1. Main isotopes used in groundwater studies.

Isotope(s)	Potential/common applications
Oxygen-18 (^{18}O) Deuterium (^{2}H) in H_2O	Origin of water, identification of recharge areas Identification of paleowater Interconnection with surface waters and between aquifers Salinization mechanisms, recycling of irrigation water
Carbon-13 (^{13}C) in HCO_3^- and CH_4	Correction for C-14 dating Identification of paleowater Groundwater dynamics Identification of methane sources
Sulphur-34 (^{34}S) and Oxygen-18 (^{18}O) in SO_4^{2-}	Identification of sources of pollution Acidification and acid mine drainage
Nitrogen-15 (^{15}N) and Oxygen-18 (^{18}O) in NO_3^-, NH_4^-, N_2	Origin of nitrates Sources of pollution Microbial denitrification processes
Boron-11 (^{11}B) in $B(OH)_4^-$ and $B(OH)_3^-$	Identification of pollution sources Characteristics of brines; source of salinity
Chlorine-37 (^{37}Cl)	Identification of pollution sources Characteristics of brines; sources of salinity
Krypton-85 (^{85}Kr)	Transport mechanisms: fissure flow Delineation of protection zones
Tritium (^{3}H)	Identification of recent recharge Water transport in the unsaturated zone
Helium-3 (^{3}He)	Dating of young groundwater
Argon-39 (^{39}Ar)	Groundwater dating
Carbon-14 (^{14}C)	Groundwater dynamics Groundwater dating
Krypton-81 (^{81}Kr)	Dating of old groundwater
Uranium-234 (^{234}U)	Dating of old groundwater, rock–water interaction
Chlorine-36 (^{36}Cl)	Rock–water interaction, dating

Most of the applications of stable isotopes of hydrogen and oxygen in groundwater studies make use of the variations in isotopic ratios in atmospheric precipitation, i.e. in the input to a hydrogeological system under study. A worldwide relation between ^{18}O of precipitation and mean annual air temperature has been observed. This dependency on temperature produces seasonal isotope variations of precipitation (winter precipitation is depleted in heavy isotopes with respect to summer precipitation), latitude effect (high latitude precipitation is depleted with respect to low latitude precipitation) and altitude effect (heavy isotope content of precipitation decreases with increasing altitude). These effects allow the use of these isotopes to delineate various hydrogeological processes as well as indicators of past and present climate changes and of palaeowaters.

The other set of tools extensively used in isotope hydrology is radioactive isotopes. Radioactive isotopes (also called radioisotopes) occurring in groundwater originate from natural and/or artificial nuclear processes. Radioactive decay of environmental isotopes makes these isotopes a unique tool for the determination of groundwater residence time (‘age’ – the length of time groundwater has been isolated from the atmosphere), which is crucial to understanding aquifer dynamics. Among the environmental radioisotopes, tritium and carbon-14 have found the widest application in groundwater studies.

The most important areas where isotopes are useful in groundwater applications include studies of recharge and discharge processes, flow and interconnections between aquifers, and the sources and mechanism of pollution. In particular, under arid and semi-arid climatic conditions, isotope techniques constitute virtually the only approach for identification and quantification of groundwater recharge. Pollution of shallow aquifers and, due to over-exploitation of superficial aquifers, also of deeper aquifers by anthropogenic contaminants, is one of the central problems in the management of water resources. Environmental isotopes can be used to trace the pathways and predict the spatial distribution and temporal changes in pollution patterns for assessing pollution migration scenarios and planning for aquifer remediation.

Furthermore, isotopes can trace dispersion and infiltration of pollutants in landfills as well as quantify degradation and migration of pollutants. Isotopes are also applied extensively to study: atmospheric processes, climate and environment changes, palaeowaters, leakages from dams and reservoirs, stream flow measurements, effluent dispersion, suspended sediment and bedload movement in ports and harbours, lake dynamics, sedimentation in lakes and reservoirs, geothermal systems, glaciology, etc.

Environmental isotopes are supplementary tools for hydrological investigations. Generally, an integrated approach, employing isotope, hydrogeological and hydrochemical data, will lead to the optimum use of these techniques and to a logical interpretation.

ROLE OF THE IAEA

The IAEA has played a crucial role in promoting and expanding the field of isotope hydrology over the last four decades. Isotope hydrology today is practised in most countries although the field began nearly 50 years ago with a few research centres in the developed countries involved in understanding the distribution of isotopes in natural waters.

The IAEA's role in building a cadre of trained isotope hydrologists worldwide is significant. In as much as isotope hydrology is an advantageous tool for the sustainable management of water resources, it is imperative that practising hydrologists are competent in the use of isotope techniques. More than 700 fellowships with an average duration of three months have been awarded over the last four decades for training at the IAEA's headquarters or other established centres. Group training events involving, national, regional and inter-regional courses of varying duration from one to eight weeks have been conducted with more than 600 participants. The trainees further improve their skills through on-the-job training associated with technical cooperation projects. Regularly undertaken Coordinated Research Projects (CRPs) provide more advanced training to a limited group of participants from developing member states. Agency-funded technical cooperation projects in isotope hydrology cater to the needs of the member states for hydrological field investigations, human resource development and strengthening of infrastructure facilities. Presently about 60 projects are active for the 2001–2002 cycle.

The IAEA maintains and provides the analytical support to the IAEA/WMO Global Network of Isotopes in Precipitation (GNIP). It periodically publishes data on the concentration of stable isotopes and tritium in precipitation samples collected at a large number of stations around the globe. This is the only database on isotopes that has been providing basic reference data for researchers in the field of hydrology and atmospheric sciences worldwide for the last four decades. Over that period of time, there has been a steady growth in the stations participating in the GNIP. The data can be found on the Internet site: http://isohis.iaea.org. Isotope monitoring of river water will provide a robust new tool for evaluating the effects of climate change and land use patterns on water resources, as well as for developing strategies for integrated watershed management. A programme has been initiated to formulate design parameters of such a Global Network for Isotopes in Rivers (GNIR).

Until recently, IAEA publications were the sole source of written material for training and education in isotope hydrology. In addition to geographical spread, the sheer number of hydrological studies with isotopes has shown a substantial increase. The number of analytical facilities has also increased steadily. A large number of laboratories in the developing countries have been established with IAEA's support.

TECHNICAL COOPERATION – THE LATIN AMERICAN CASE

Technical cooperation (TC) with the member states increasingly promotes tangible socio-economic impact by contributing directly in a cost-effective manner to the achievement of the major sustainable development priorities of each country. One growing emphasis in TC activities is helping countries and regions to investigate and manage water resources using isotope hydrology. The IAEA assistance aims at developing experience and expertise through training, expert advice and the provision of equipment to improve local infrastructure and build capacity to study water resources using tracer isotopes. Over the last decade, the IAEA has supported some 160 TC projects, amounting to US$18.8 million.

Isotope hydrology TC in Latin America started in the 1960s. Various projects were undertaken in collaboration with United Nations Development Programme (UNDP) in Chile, Colombia, Ecuador, Guatemala, Uruguay and Mexico. Since then about 100 TC projects have been successfully completed. A majority of the projects undertaken in Central America were related to geothermal and surface water studies, whereas in Latin America focus was on groundwater projects (Figure 1).

Twenty-one regional and inter-regional group training events have been conducted in Latin America since 1980 with more than 200 participants. The IAEA coorganized the II Spanish-American course on Groundwater with UNESCO and IMFIA-University of

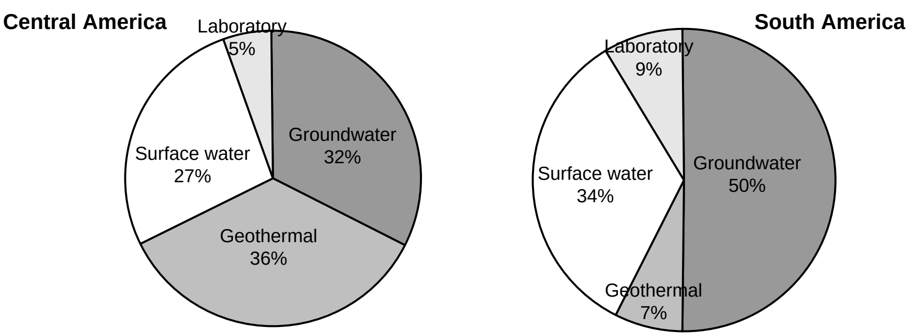

Figure 1. Distribution of TC projects in Central and South America.

Montevideo. The trainees further improved their skills through on-the-job training associated with technical cooperation projects. The availability of trained scientists with competence in isotope hydrology has improved the implementation of technical cooperation projects by interfacing indigenous capability with outside expertise and developing capacity for related research.

Two major regional programmes have been recently initiated in Southern America: RLA/8/031 and RLA/8/035. The objective of project RLA/8/031 on the 'sustainable management of groundwater resources is to study groundwater availability, quality and pollution risks in order to optimize the sustainable management of groundwater resources in the region, as a multidisciplinary and inter-institutional activity, and to facilitate the regional cooperation of these activities. This project is being implemented by Chile, Colombia, Costa Rica, Ecuador, Paraguay, Peru and Uruguay. Traditional hydrological tools, combined with nuclear techniques, are applied to investigate aquifers in the region.

The regional project, RLA/8/035 is fully integrated to a GEF/OAS/WB project on 'environmental protection and sustainable development of the Guarani Aquifer system'. The main objective of this project is to support Argentina, Brazil, Paraguay and Uruguay in jointly elaborating and implementing a common institutional framework for managing and preserving the Guarani Aquifer for current and future generations.

Within the framework of a four-year regional TC project, the IAEA will provide funds and support for the use of isotopes to delineate the extent and character of the aquifer, and through these activities, contribute to building capacity and strengthening institutions in the countries of the region. Isotope data provide a critical input for strengthening the conceptual model of an aquifer system, which in turn underpins decision-making in resource management and environmental protection.

CONCLUSIONS

The IAEA has played a pivotal role in promoting and expanding the field of isotope hydrology over the last four decades. Now greater efforts are needed to impart knowledge and skills to a new generation of hydrologists. Steps for inducting isotope hydrology course in university curricula have been initiated. An additional avenue for promoting isotope hydrology education at university level is to sponsor chairs in isotope hydrology at selected universities. This sponsorship would provide a higher profile to the awardees and enable them to serve as a magnet or focal point for national or regional educational and applied research activities.

In addition to joint publications by the IAEA and UNESCO (2001), a CD-ROM is in preparation as a teaching and self-learning tool. Continuing activities in the IAEA's isotope hydrology programme facilitates developments through research, information dissemination, education and training, and support for field applications including analytical services.

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CHAPTER 5

The use of isotopic techniques in determining groundwater pollution vulnerability – A Latin American perspective

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ABSTRACT: The degree of vulnerability of groundwater to pollution can be defined as a group of physical, chemical and biological characteristics of the unsaturated zone and/or aquitard, which control the susceptibility of the aquifer to degradation by contaminating events of anthropic origin. Vulnerability is thus dependent upon two factors: i) the degree of access, in hydraulic terms, provided for penetration of contaminants and ii) the attenuation capacity offered by the geological medium. Isotopic techniques are useful in determining vulnerability to the extent that they allow for a better understanding of these two particular factors. The possibility of dating groundwater is a powerful tool in estimating hydraulic access to the aquifer. In the same way, nitrogen and carbon isotopes are able to provide information regarding such contaminant attenuation processes as the denitrification and biodegradation of halogenated solvents, respectively. This work shows the way in which some of these techniques can assist in determining vulnerability, based on examples in Latin America.

INTRODUCTION

Since the 1980s, there has been a period of intense development of mapping techniques for determining the degree of vulnerability of aquifers to anthropic contamination. More recently, aquifer vulnerability has served as a basis for many groundwater quality control programmes in various countries, principally those in Europe. This has come about due to the fact that it is more economical and less harmful to the environment to control potentially contaminating activities, using different levels of aquifer vulnerability as a basis, than to attempt to apply universal control over all such activities in a generalized manner (Foster et al., 2002).

In fact, it is difficult to represent the complexity of aquifer systems and the subsurface behaviour of contaminants in a complete manner through the use of a simple vulnerability method. As a result, this has led some hydrogeologists to advocate that it would be more realistic to analyse each case of contamination individually. However, such a procedure is confronted by human resources and economic limitations, especially in developing countries such as those of Latin America.

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Many of these problems can be overcome if it is made clear that the mapping of vulnerability is not a substitute for detailed study, but a first step in the identification of the most significant environmental hazards on a state, provincial or even municipal scale. In this way, vulnerability zoning may be able to assist in the identification of action priorities on the part of environmental control agencies, not only indicating those activities that have already been installed and which may be endangering the aquifer, but also defining the environmental requirements to be met by new activities, based on groundwater protection and support capacity.

The correct application of vulnerability techniques should be preceded by an understanding of a conceptual groundwater circulation model, principally those mechanisms that control aquifer recharge. There is an unquestionable relationship between recharge and vulnerability, since the transport of many contaminants to the saturated zone of the aquifer occurs in the dissolved phase, as part of the recharge process (Foster, 1998). In this process in particular, isotopic techniques, with emphasis on environmental isotopes, have been found to be widely applicable, as they allow for identification of the origin of the water that is infiltrating the aquifer.

If, on the one hand, the zoning of land is clearly important as the best way of occupying it (and vulnerability methods can be a useful tool for this purpose), there are still several technical aspects that need to be studied in depth and resolved. One of these aspects is the need for studies that validate the results of different methods of determining vulnerability. Once again, isotopic techniques can be of great importance, to the extent that they measure the time of arrival of water at the saturated zone of the aquifer and, in some cases, even provide indication of the origin of this water and the path it has followed through the rocky medium.

THE CONCEPT OF AQUIFER VULNERABILITY

The concept of aquifer vulnerability was introduced by Le Grand (1964) in the USA, Albinet & Margat (1970) in France and, to a wider extent in the 1980s by various other authors (Aller et al., 1987, Bachmat & Collin, 1987, Foster & Hirata, 1988). Since then, this concept has been used in order to express the following:

- intrinsic characteristics of strata that separate the saturated zone of the aquifer from the surface, which determine the level of sensitivity to adverse effects caused by the imposition of a contaminant load of anthropic origin;
- classification of aquifers based on the socio-economic importance of the water resource at the present time and in the future, including the possibility of being replaced by other sources; and
- classification based on the importance of the aquifer in maintaining important ecological areas.

From a technical scientific point of view, the first definition, with some modifications, is that which is most widely used.

Basically, it is possible to gain an understanding of the degree of vulnerability of an aquifer based on the following (Foster & Hirata, 1988):

- degree of hydraulic access regarding contaminant penetration on reaching the saturated zone of the aquifer (advection of contaminants); and

- attenuation capacity of the layer that covers the saturated zone, resulting from the dispersion, retention or physical–chemical reaction of contaminants (retardation and degradation).

Along the same lines, the authors define the hazard of groundwater pollution as the probability of those more surficial portions of the aquifer being degraded by some kind of surface activity. In this way, the hazard of contamination of an aquifer is understood as an interrelationship between the vulnerability of the aquifer and potential contaminant load.

It is possible to mention a large number of vulnerability mapping techniques and their application in a variety of hydrogeological contexts (Vrba & Zaporozec, 1994; Hirata & Rebouças, 1999). In Latin America, the most common methods are as follows: GOD (Foster & Hirata, 1988), DRASTIC (Aller et al., 1987) and SINTAC (Civita et al., 1990).

An analysis of the various existing methods of determining vulnerability shows that there are some problems.

- Although several authors, including two major professional working groups (NRC, 1993; Vrba & Zaporozec, 1994), have coined a few definitions, which, in a tentative manner, would unify the concept of vulnerability and its application, there is still no consensus of opinion among hydrogeologists regarding this subject (Foster et al., 2002). One of the most notable points is that several methods are restricted to analysing vulnerability as the arrival of contaminants at the saturated zone, in other words, the extent to which the material between the surface of the soil and the saturated zone of the aquifer allows a contaminant to cross it. Besides this, other methods also include consideration of the degree of mobility of contaminants in the saturated zone. This, however, does not appear to view vulnerability mapping from the most useful perspective, namely that of providing a framework for planning and controlling activities at on the surface of the land.
- The lack of a widely accepted concept makes it difficult to compare different methods, principally due to the fact that almost all of them result in relative degrees of vulnerability. An absolute index of aquifer pollution vulnerability is far more useful for all practical applications in land-use planning and effluent discharge control. An absolute index can be developed provided each class of vulnerability is clearly and consistently defined (Table 1). In this way it is possible to overcome most (if not all) the common objections to the use of an absolute integrated vulnerability index as a

Table 1. Practical definition of classes of aquifer pollution vulnerability (Foster et al., 2002).

Vulnerability class	Practical definition
Extreme	Vulnerable to most water pollutants with relatively rapid impact in many pollution scenarios
High	Vulnerable to many pollutants, except those highly absorbed or readily transformed, in many scenarios
Medium	Vulnerable to some pollutants, but only when continuously discharged/leached
Low	Only vulnerable to conservative pollutants in long term when continuously and widely discharged/leached
Negligible	Confining beds present with no significant vertical groundwater flow

framework for groundwater pollution hazard assessment and protection policy formulation (Foster et al., 2002).

- The different methods have shown themselves to be insufficient for assessing vulnerability in areas of complex geology, for example in crystalline bedrock aquifers (fractured flux), recent volcanic rocks, multi-layer formations, karstic aquifers and dual-porosity systems. The highly heterogeneous nature of the material, resulting in the difficult prediction of pollutant flux along preferential pathways, has had the effect of making existing methods simplify and generalize hydrogeological aspects and lithological groups, often without the necessary detailing of the aquifer medium. The methods fail to incorporate geological concepts that describe preferential fluxes in a more suitable manner. For example, in fractured aquifers of the kind found in ancient formations, there should be analysis not only of the tectonic event that produced the fracturing, but also of those that maintain some fractures open and others closed, factors normally associated with the most recent phase of deformation.
- Although there are a few studies that include a critical analysis of existing methods, most importantly DRASTIC (Holden et al., 1992; Bates et al., 1993; Kalinski et al., 1994; Rosen, 1994), the different techniques involved still lack validation based on detailed studies in the field with the systematic monitoring of groundwater quality. An analysis of various contamination cases, including a range of different anthropic activities, would provide the necessary basis for this validation. In the same way, this kind of study would assist in better defining an absolute vulnerability index, or at least a system of equivalence between the different methods.

HOW ISOTOPIC TECHNIQUES ASSIST IN THE CHARACTERIZATION OF AQUIFER VULNERABILITY

There are basically two factors that control the vulnerability of an aquifer, namely the degree of hydraulic access and the protection that surficial layers offer the saturated zone of an aquifer (non-saturated zone and/or aquitard) in the form of attenuation capacity.

Table 2. Hydrogeological factors controlling aquifer pollution vulnerability.

Component of aquifer pollution vulnerability	Hydrogeological data	
	Ideally required	Normally available
Hydraulic accessibility of the saturated aquifer to penetration of pollutants	• degree of aquifer confinement • depth to groundwater table or groundwater strike • unsaturated zone moisture vertical hydraulic conductivity of strata in vadose zone or confining beds	• type of groundwater confinement • depth to groundwater table or top of confined aquifer
Attenuation capacity of strata overlying the saturated zone, resulting from the physicochemical retention or reaction of pollutants	• grain and fissure size distribution of strata in vadose zone or confining beds • mineralogy of strata in vadose zone or confining beds	• grade of consolidation/ fissuring these strata • lithological character of these strata

Normally, vulnerability assessment methods are limited by a lack of hydrogeological information (Table 2). Isotopic techniques can be important tools to the extent that they assist in providing part of this information (Table 3).

It is only possible to develop satisfactory vulnerability mapping based on adequate characterization of a conceptual model of aquifer flow and, most importantly, determination of its recharge area.

Table 3. The use of isotope techniques for groundwater pollution vulnerability characterization.

Component of aquifer pollution vulnerability	Information related to	Isotope techniques
Hydraulic accessibility of the saturated aquifer to penetration of pollutants	Identification of recharge/ discharge areas Degree of aquifer confinement Definition of period of recharge Identification of palaeowater Interconnection of surface and groundwater	^{18}O* in H_2O ^{2}H* in H_2O ^{85}Kr flow through fissured rock ^{3}H in H_2O (environmental and applied tracer) ^{51}Cr, ^{58}Co, ^{60}Co, ^{131}I (applied tracers)
Attenuation capacity of strata overlying the saturated zone, resulting from the physicochemical retention or reaction of pollutants	Determination of transit time of water through the saturated and unsaturated zones (advective flow)	Dating of young water ^{3}H ^{3}He Dating mid-age water ^{39}Ar ^{14}C, correction with ^{13}C in HCO_3^- ^{81}Kr Dating old water ^{81}Kr ^{234}U
	Identification of physicochemical reactions of pollutants (degradation)	^{15}N and ^{18}O in NO_3^-, NH_4^+, N_2, microbial denitrification processes ^{13}C, biodegradation of synthetic solvents and fuel ^{34}S, oxidation/reduction of sulfur
Pollution hazard factors	*Information related to*	*Isotope techniques*
Definition of pollutant origin	Identification of source of pollution	^{34}S and ^{18}O in SO_4^{2-}, acidification and acid mine drainage ^{37}Cl, ^{11}B, in $B(OH)_4^-$ and $B(OH)_3^-$ characteristics of brines; sources of salinity ^{15}N and ^{18}O in NO_3^-, NH_4^+, N_2 ^{18}O, ^{2}H in H_2O, salinization mechanisms, recycling of irrigation water ^{14}C and ^{13}C for origin of sources associated to organic compounds

Tracers can be utilized in estimating recharge, these being categorized as environmental (already present in the geosphere) and applied tracers (injected by the researcher). According to Lerner (1990), there are several ways that tracers can be used:

- signature methods, in which particular parcels of water are labelled and traced;
- throughput methods, when fluxes of tracer and water are calculated in the unsaturated zone, usually for environmental tracers; and
- turnover or transit time calculations, which are used for whole aquifers, usually with environmental tracers.

Among the environmental isotopes, ^{18}O and ^{2}H are those most widely used in defining recharge areas or their origin. These isotopes are affected by isotopic fractionation processes, in other words, changes in the isotopic relationships between $^{18}O/^{16}O$ and $^{2}H/H$ during the processes of evaporation and condensation of water (liquid–vapour phase exchange). This phenomenon labels the water and allows its origin to be defined, as it is a function of altitude, distance from the coastline and rainfall dynamics in a given area. Transpiration by plants does not affect this fractionation process.

An example of the use of these isotopes is the characterization of the recharge area of the Colima Aquifer, located in the Valle Central region of Costa Rica. The aquifer in question is one of the most important water sources supplying the population of San José. By means of geological and isotopic evidence, it was possible to determine that recharge of the Colima Aquifer was principally associated with regions of high elevation and did not come about as a result of the direct infiltration of water from other aquifers nearer the surface (Figure 1). At the present time, there is increasing occupation of these highland areas particularly for agricultural purposes with the extensive use of nitrogenated fertilizers. The rapid movement of recharge waters in this kind of aquifer comprised of recent volcanic rock, confirmed using tritium dating techniques (BGS/SENARA, 1988), shows that this is a vulnerable aquifer. Based on the evidence gathered, it is recommended that the occupation of these highland areas be conducted in a careful manner, with a view to protecting the whole Colima Aquifer. Particularly in this area, there should be definition of effective vulnerability mapping of the aquifer system.

Groundwater dating is a powerful tool in identifying groundwater transit time until its arrival at the saturated zone. Recent water present in an aquifer may indicate rapid contaminant access to its saturated zone. This would indicate a lower level of protection, as the degree of degradation is, in the majority of cases, proportional to transit time through the geological medium.

Isotopic dating techniques measure the time between recharge and collecting a sample from the aquifer. The presence of tritium, an isotope showing rapid radioactive decay (12.42 years), is confirmation of recent water and may be indication of a vulnerable aquifer, at least from a hydraulic point of view. On the other hand, the dating of water that has been present for dozens or even hundreds of years is an indication of a well-protected aquifer. In the same way, the absolute dating of water is an excellent tool for confirming the degree of confinement to which the aquifer is subjected.

Solis & Araguás (1994) identified that the recharge of aquifers in the Valle de Cochabamba region (Bolivia) is associated with areas lying close to alluvial fans. Results obtained using ^{14}C and tritium made it possible to identify at least three aquifer zones. These are as follows: the first of recent age, with a tritium content equal to that observed in rainwater (4–6 UT), followed by a transition zone (<2 km in width) where the tritium

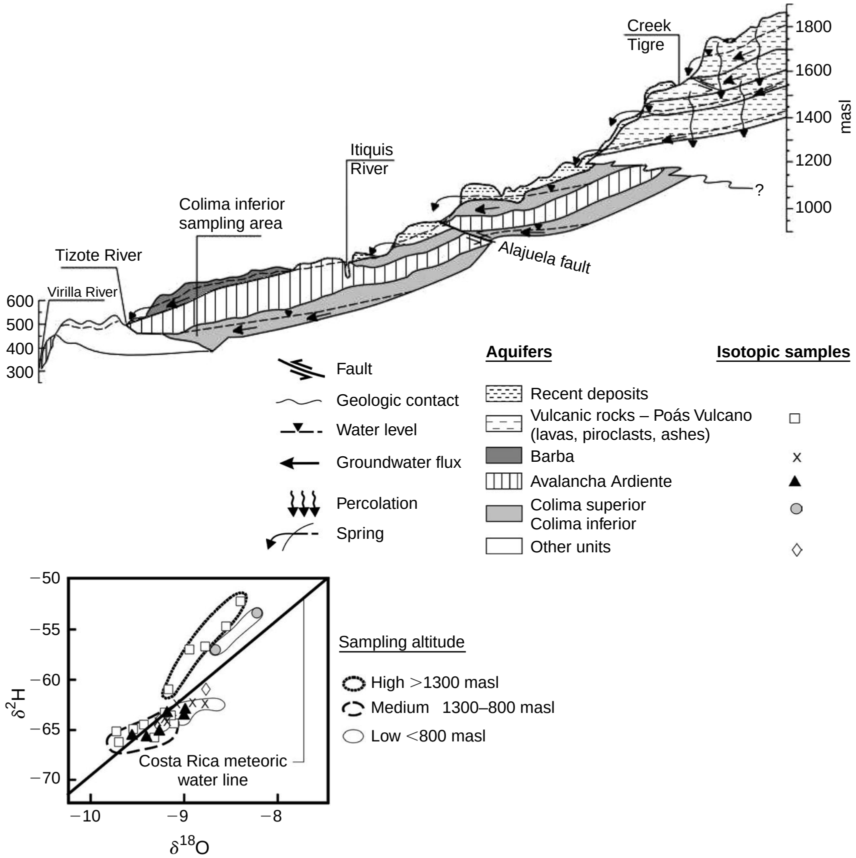

Figure 1. Isotopic groundwater evolution in Valle Central (San José, Costa Rica).

concentration drops to 0 UT, and a third, where the radiometric age derived from radiocarbon content in the water is in the order of 10,000 to 15,000 years. In this third zone, Stimson et al. (1992), also studying the dynamics of environmental isotopes, showed that recharge waters would have occurred in a colder climate than that experienced at the present time. In addition, the study identified the important contribution made by contaminated surface water to the aquifer, thus endangering groundwater quality. These studies allow for definition of those areas of the aquifer that are most vulnerable, as well as future impacts on the groundwater resource resulting from careless occupation of the land.

Sometimes it is possible to estimate transit time in the unsaturated zone through the analysis of environmental tritium in interstitial water or even by means of the injection of tritium itself. The studies carried out by Araguás et al. (1994) on agricultural land of volcanic origin in Costa Rica is a good example of estimation of the average rate of transport of a non-reactive solute (tritium) through the vadose zone profile and quantification of aquifer recharge.

However, assessments of contaminant transport through the use of environmental and artificial tracers have been shown to have limitations in dual-porosity formations. Unfortunately, there are no studies in Latin America regarding this particular problem.

Through the accompaniment of environmental tritium profiles (1.0 m/a), research carried out in Great Britain on the Chalk Aquifer has shown that, although there was very little possibility of contamination, there was intermittent groundwater contamination with faecal bacteria in deep (10–25 m) water table areas (Foster & Smith-Carington, 1980). A possible explanation for this problem was the presence of a fissure flow in a dual-porosity system where the velocity of water movement was high enough to prevent full interchange of solutes with the microporous matrix (Barker & Foster, 1981).

Another important factor in studying vulnerability is the assessment of the degree of degradation of contaminants. With regard to some contaminants, it is possible to estimate this through analysis of their isotopic content. This is especially so in the case of nitrate, but other pollutants can be analysed in the same way, for example synthetic organic compounds (Stahl, 1980), petroleum hydrocarbons and sulphur (Clark & Fritz, 1997).

Denitrification is the biochemical reduction of nitrate, which occurs in low oxygen concentration environments, producing nitrogen gas (N_2) as an end product. This chemical transformation, normally coming about through the mediation of bacteria, has been considered as a single-step and unidirectional reaction process, which leads to the isotopic enrichment of NO_3^- and which follows the Rayleigh fractionation model (Mariotti et al., 1988).

Dilution of the contaminant plume, which may sometimes be confused with denitrification, does not alter the isotopic relationships of nitrogen present in the nitrate. This would only occur if both waters in the mixture have different isotopic compositions. In this case, it would be possible to arrive at an estimate of this process through binary mixture models, where isotopic composition evolves in a hyperbolic manner, as a function of concentration (Mariotti et al., 1988).

Finally, isotopes can assist in determining the contamination source, based on its isotopic signature. This would allow an estimation to be made of the degree of hazard of contamination (interrelationship between vulnerability and contaminant load) and would also assist in studies carried out with a view to the validation of vulnerability methods. Isotopes are particularly useful in identifying contamination sources involving nitrate, synthetic organochlorinated compounds (Van Warmerdam et al., 1995), sulphur (Krouse, 1980) and chloride (Clark & Fritz, 1997).

For example, the origin of contamination by nitrates can be determined through the relationship between $^{15}N/^{14}N$. Typical $\delta^{15}N$ values for nitrified soil organic nitrogen fall within the $+4$ per mill to $+9$ per mill range; for nitrogenous fertilizers these values lie between -4 per mill and $+4$ per mill; and for animal and sewage wastes, $> +10$ per mill (Heaton, 1986). These figures may undergo alteration and overlap in accordance with the characteristics of the nitrate generating process. Based on water samples taken from the aquifer, the results can indicate, depending on the $\delta^{15}N$ relationship and evidence gathered in the field, the origin of the contamination.

The replacement agricultural areas formerly used for coffee cultivation by urban developments with no adequate sewage collection network has been one of the causes of an increase in nitrate concentrations in the water in aquifers in the Valle Central region of Costa Rica. Using nitrogen isotopes, Reynolds et al. (in preparation) demonstrated a relationship between the $\delta^{15}N$ value, typical of animal origin ($> +10$ per mill) and soil use in the capture zones of monitored wells.

Studies analysing the origin of the salinization of aquifers in Latin America can be found in Panarello et al. (1994), who studied the La Plata region (Argentina) and in

Maldonado et al. (1994), who utilized environmental isotopes in aquifers of the River Guayas delta (Ecuador).

CONCLUSIONS

In the last few years, techniques for determining groundwater pollution vulnerability have been used in an ever-increasing manner in groundwater protection and management programmes in Latin America. Although some of these techniques are more than 25 years old, there are still some problems that should be resolved with a view to more widespread and reliable use of the methodology involved, including: i) better definition of the concept of aquifer vulnerability; ii) development of methods capable of establishing vulnerability in such areas of complex geology as karstic, fractured, semi-confined and double-porosity aquifers; and iii) validation of vulnerability methods, by means of field studies, together with detailed monitoring.

Although isotopic techniques are important tools in determining the hydrodynamics of aquifers, they are still little used in Latin America. One of the main problems in mapping aquifer vulnerability is a lack of hydrogeological information, principally associated with the determination of the recharge zone and time of arrival of water at the saturated zone. In this respect, isotopic techniques have been of great assistance.

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Section 3

Papers from technical sessions

CHAPTER 6

Arsenic in groundwater: Its impact on health

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ABSTRACT: Arsenic is a component of groundwater, which becomes toxic for human health when high levels are reached. The most important disease is regional endemic chronic arsenic poisoning. Its long-term effects on health are skin lesions, skin cancer and internal cancers of the bladder, kidney and lung. In Argentina, the central and northern regions, referred to as the Chaco-Pampeana Plain, are the most affected by this disease. Arsenic in groundwater is due to the presence of volcanic ash in the loess sediments of the region. A study case of the impact of arsenic in groundwater on the health of the population in the northeastern Mendoza Province of Argentina is presented here. Preventive measures, including alternative sources, deeper wells, education to raise awareness on health risks, multidisciplinary work and joint actions between health and water managers, are necessary to supply safe water to the population.

INTRODUCTION

Although usually in small amounts, arsenic is present in most waters. This issue is of great interest in countries such as Argentina, Chile, Brazil, Peru, Bolivia, Mexico, Thailand, Bangladesh, China, India and the United States, where severe cases of contamination have taken place.

In Bangladesh, increased levels of arsenic in groundwater were identified after the appearance of arsenic-related diseases. The scale of the problem was calculated based on the poisonous effects diagnozed in the population. In several world regions, hydric diseases connected with the presence of arsenic have been described, such as in Antofagasta (Chile), Minas Gerais (Brazil), Salvador de Arriba and Coahiula (Mexico), Formosa Province (Taiwan) and perhaps in the Puna of Atacama (Peru & Bolivia) (Sai Siong Wong et al., 1998).

The toxic effects of arsenic have been known since ancient times and their consequences on human health have been studied since the nineteenth century. The World Health

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Organization finally fixed the maximum recommended limit of 0.01 mg/l. Arsenic tolerance guidelines vary from one country to another. In the United States and Russia, for instance, arsenic content in drinking water above 0.05 mg/l is considered dangerous, the European Union considers it so above 0.01 mg/l. Argentina has fixed the maximum limits of arsenic concentration in water of 0.05 mg/l, above that of some other countries, and even so, there are endemic areas where these values are notoriously exceeded.

The solution to arsenic contamination of water in developing countries is not simple because of their socio-economic and facility problems. It is therefore necessary to introduce preventive measures to avoid health risks in the population.

ARSENIC ORIGIN AND USE

Naturally occurring arsenic is found in common minerals such as arsenopyrite (FeAsS), rejalgar (As_2S_2), oropiment (As_2S_3) and arsenic trioxide (As_2O_3); it is found occasionally found as a pure element.

Undoubtedly, arsenic is part of history; having been mistaken for sugar or flour, it was used as a powerful poison. It has been known under names that speak for themselves, such as 'the king of the poisons', 'succession powder', etc. Murderers have used its ability to slowly kill their victims, apparently of natural causes. They used high doses of arsenic, – higher than those appearing in water – which produced quick deterioration in health and ultimately death. Low exposures, such as intake from water, also have detrimental effects in the medium and long term.

Arsenic compounds have different industrial and agricultural applications; unless proper precautions are taken, workers can be greatly exposed to this element. The most common use is in insecticides (arsenate of copper and calcium), fungicides, herbicides and defoliants. It is also used in the manufacture of glass and poisonous gas (lewisit and adamsit) and as a pigment in fireworks and paints (arsenic disulphide). Until the introduction of penicillin, the trivalent organic arsenic Neosalvarsan was the most sensitive treatment against syphilis and Fowler liquor (arsenate of copper and potassium) was used for psoriasis treatment.

Currently, there are several cases of arsenic intoxication resulting from the consumption of water with high arsenic content.

Arsenic contamination in aquifers can be related not only to the incorporation of contaminants from the surface as a consequence of human activity, but also to the natural processes of rock–water interaction, caused by specific geological environments. The most important chemical species in natural waters are H_3AsO_3, $H_2AsO_4^-$ and $HAsO_4^{2-}$, highly soluble and stable. Arsenates are more toxic and are found in reductor environments and under oxic conditions.

It is crucial for social communities to know about the level of arsenic in waters for human consumption. Nowadays, this issue is considered a worldwide problem due to its toxic effects on health.

DISEASES CAUSED BY ARSENIC

It has been determined that one in ten people who drink water with an arsenic content above 0.50 mg/l die of lung, gall bladder or skin cancer. Occupational exposure, mainly in cases of inhalation, increases the risk of lung cancer when cumulative levels of $0.75\,mg/m^3$ are reached. This amount corresponds to 15 years of exposure in a room with a concentration scale of $0.050\,mg/m^3$ (Stanwell Smith, 2001).

The arsenic content of water and food enters the body by mouth. After being absorbed by the blood stream and distributed to the different organs, arsenic deposits mainly in the liver, kidneys, heart, lungs and skin. Hair and fingernail deposits form two weeks after its administration and remain there for years. Inorganic components are eliminated by urine. Other less significant ways of excretion are in faeces, perspiration, breast milk, skin, hair and fingernails (Goodman & Gilman, 1991).

There are regions where continuous and prolonged consumption of water with high arsenic content produces skin and visceral manifestations referred to as HACRE (Chronic Endemic Regional Hydroarsenicism). It is believed that when water contains above 0.5 mg/l of disodic arsenic or above 0.12 mg/l of metalloid arsenic, it can produce chronic intoxication. HACRE has characteristics of its own, different, taken as a whole, from those of other acute, sub-acute and chronic arsenic intoxications. To catch the disease, a consumer must have drunk water with a dangerously high arsenic content for months or years. Both men and women are affected; although it is more frequent in men with a ratio of 4:1. Most patients are adults, children being an exception. The clinical manifestations of HACRE include skin and mucous lesions and visceral cancers (Grisnpan, 1985; Tello, 1981).

Palmo-plantar keratoderma is the most frequent symptom in the diagnosis, affecting as many as 96 per cent of patients. It appears after one to 15 years of drinking water with high arsenic levels. Palmo-plantar localization is bilateral and almost simultaneous. It may be localized or diffuse and is generally verruciform. Sometimes it may make closure of the hand difficult. Skin gets thicker, becomes dry and rough, and is of a greyish yellow or blackish grey colour.

Skin tumours are the second symptoms in order of frequency. The number of carcinomas varies; generally they are multiple, placed in the trunk, mostly in the back, chest, and in the limbs and scalp. There are more lesions in covered areas than in exposed parts, showing that the sun, if present, does not play a role in HACRE. The appearance of malignant tumours depends on the time of arsenic exposure – Bowen's disease within ten years of exposure and invasive cancers after 20 years (Ray Bettley & O'Shea, 1975; Ohnishi et al., 1997; Evans, 1977).

Melanoderma is a dark grey or black pigmentation, which may be diffuse or circumscriptive. It is the least frequent skin manifestation (22 per cent of patients). Most frequently, it appears in the trunk and in the root of the limbs.

Oral and laryngeal mucous leukoplakias predispose patients to cancer of the larynx.

In visceral cancers, i.e. internal organs, 30 per cent of HACRE patients die from internal neoplasia and 33 per cent from lung cancer. The affected organs, according to a frequency scale are, lung, stomach, larynx, esophagus, pharynx, trachea, liver and breast. Carcinomas take from 20 to 30 years to develop, depending on the arsenic intake.

HACRE evolution is slow and progressive. Once initiated, the process does not disappear even if the person moves to an area with water containing low levels of arsenic. The prognosis is characterized by the development of visceral cancers.

Skin lesions can be treated with medication, cryosurgery, electro-coagulation, radiotherapy or classic surgery.

OCCURRENCE OF ARSENIC IN ARGENTINA

In Argentina the first pathologic manifestations occurred early in the twentieth century and were known as 'Bell Ville disease', after the city where the greatest number of cases took place. With treatment and better management of the disease in different urban centres it became apparent that the affected area was bigger, encompassing Buenos Aires, Córdoba, La Pampa, Santa Fe, Santiago del Estero, Chaco, Salta, Tucumán, Catamarca, Formosa and San Luis Provinces (Biagini, 1975; Baliña et al., 1981).

Across the whole Chaco-Pampeana Plain (central and northern Argentina), several authors traced the origin of arsenic in the groundwater to the presence of ash and volcanic glass in the loess sediments of the region (Nicolli et al., 1997; Smedley et al., 2000).

At La Pampa Province, groundwater is the main source of water supply. Most urban areas count on water service providers. However, wide areas in this province face slim chances of water supply because of its high arsenic content, exceeding in some cases 7 mg/l and therefore being unsuitable for human consumption. Exploitation is highly restricted and the delimitation of high-arsenic content areas represents a certain degree of complexity since arsenic's behaviour is highly heterogeneous and varies depending on whether it is considered in its horizontal or vertical distribution, and on its relationship with geomorphology, hydraulics and lithology. Lack of medical knowledge regarding the physico-chemical characteristics of the water in the region and in its nearby areas is common. Besides, the recommended limits for human consumption are generally ignored and therefore so are the risks and consequences this has to health. Perhaps, this is all due to the fact that there are no reliable sanitary statistics on pathologies registered and certified by health authorities (Schulz et al., 2002).

At Catamarca Province, surveys were conducted revealing high contents of arsenic in the water in the Departamento de La Paz and the Departamento de Antofagasta de la Sierra, where groundwater arsenic levels ranged from 0.01 mg/l to 0.17 mg/l.

In the central-northern region of Buenos Aires Province arsenic content is frequently found to exceed 0.05 mg/l. In the southwest, arsenic concentrations reaching values of as much as 0.15 mg/l appear anarchically and frequently independently of the salinity and depth of the aquifer level exploited. Spatial hydrochemical variations could be related to the transport conditions and deposition of volcanic ashes, and to the type, chemical composition and age of the eruption as well as to the weather conditions of each area. (Carrica & Albouy, 1999).

In the southern Córdoba Province (Cabrera & Blarasín, 2001) arsenic content of as much as 0.5 mg/l was found in the phreatic aquifer composed of quaternary eolic sediments.

At the northeastern Tucumán Province, where the main provincial economic activity takes place, arsenic values of 1.6 mg/l were detected in the groundwater (Nicolli et al., 2001).

At the Departamentos de la Banda y Robles, Santiago del Estero Province, cases of hydroarsenic poisoning have been reported since 1983 and deaths were caused by values of 1.00 mg/l of this oligo-element in groundwater, i.e. 100 times the limit recommended by the World Heath Organization. The level of the water table rose as a consequence of the increase in risk in 1970. There is a horizon of volcanic ash between 2 and 3.50 m in depth and the amount of arsenic in the groundwater has been growing as these ashes come to form part of the saturation zone. This creates an area of very high risk of catching the disease, primarily for those consuming the groundwater contaminated with arsenic. Therefore the Public Health Department of the Province was informed of the outcomes of the hydrogeological surveys and launched a campaign to stop hydroarsenic poisoning. Hair and fingernail samples were collected from the population and four cases of the disease were detected. A sanitary education campaign was then conducted and water started being supplied by tank trucks as an immediate solution, until the most appropriate ones of deep perforations, aqueducts or reverse osmosis could be adopted. (Herrera et al., 1999).

Mendoza Province: a study case

At Mendoza Province, arsenic was detected in groundwater in an important region located at the northwest end of the north oasis, encompassing a great part of the Departamento Lavalle (Alvarez, 1985, 1993). Arsenic concentrations ranged from 0.01 mg/l in the south to 0.22 mg/l in the northeast and northwest, showing no difference in the various exploited aquifers (Figure 1).

The hydrochemical surveys performed reported that the volcanic ash found in alluvial sediments of several hydrographic basins located on the eastern hillsides of the pre-Cordillera were responsible for the presence of arsenic in groundwater. These sediments are interbedded with alluvial plain sediments, situated away from the alluvial cones of the Mendoza River. Groundwater in this region is very important as it is the hydric reserve supplying the oasis during droughts; besides that, in some cases, it is the exclusive source for crop irrigation, and it is the only source for human consumption (Figure 2).

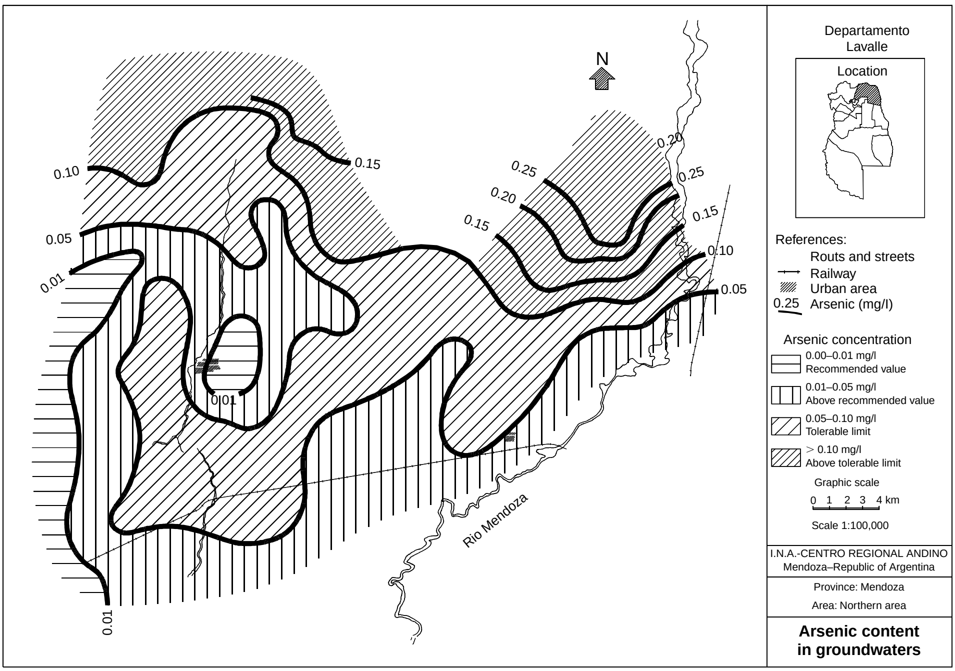

Figure 1. Arsenic content in groundwater in northeastern Mendoza.

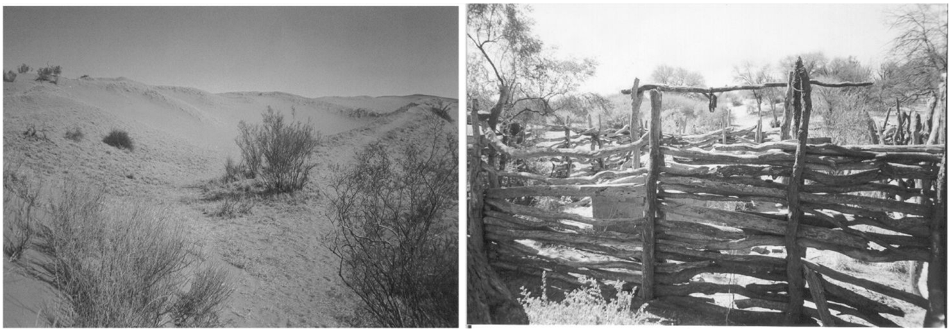

Figure 2. Typical landscape and domestic well in the northeastern Mendoza Province.

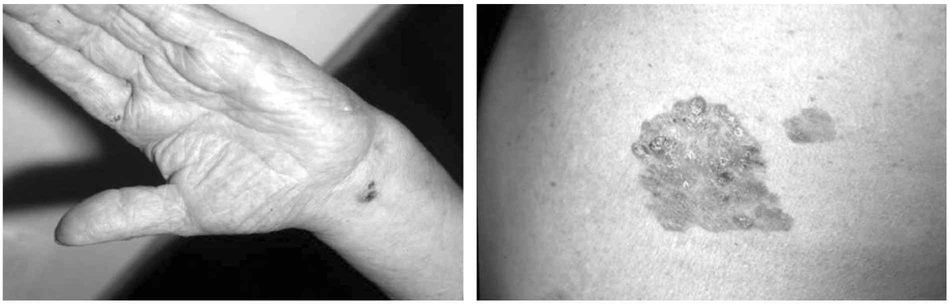

Figure 3. Clinical photographs show multiple palmar-plantar arsenic keratosis and multiple Bowen's disease on the trunk and hand.

The rural school situated at San José has a perforation that yields 0.18 mg/l of arsenic in the water, which has been used to supply the school for a long time now. In spite of the fact that this has been known for a while, no studies have been performed on the effect this might be having on health, nor have possible consumers been alerted or measures been taken to provide them with arsenic-free water.

In September 2002, samples were taken from four domestic wells, extracting water from the most superficial aquifer, and from a 280 m deep perforation, the Posta San Gabriel, which supplies water not only to many families in the area, but to a school and to a police station. The latter perforation yielded a 0.67 mg/l arsenic content, whereas the domestic well content ranged from 0.11 to 0.79 mg/l.

Regarding sanitary conditions, it was possible to gather information from three patients with hydroarsenic poisoning in the Departamento Lavalle, (Mendoza). Two of them died. The first victim was 59 years and suffered from HACRE and lung cancer; the second died at the same age, a victim of hydroarsenic poisoning, lung cancer and a spinocellular carcinoma of the skin.

The patient still alive has lung cancer and started radio- and chemotherapy treatment five years ago. This patient presents with multiple palmar-plantar arsenic keratosis and multiple Bowen's disease in the trunk and limbs (Figure 3).

These three individuals lived in an endemic area until they were about 20 years and drank well water of high arsenic content from La Chilca and El Ramblón de las Cabras, (0.67 and 0.79 mg/l respectively).

This area, bordering San Juan Province near the Desaguadero River, shares the high arsenic values found in the city of El Encón. This city has a very small, stable population, which is supplied with water treated via inverse osmosis so as to eliminate the arsenic content.

Currently, this Mendoza area is involved in an incipient livestock activity. It houses approximately 50, mostly large, families, composed of young people subjected to a high risk of catching this life-threatening disease over time. It is essential and urgent that systematic and multidisciplinary research of the current situation is conducted so that a solution can be found and these families do not have to leave – an unwelcome option to those who are native to the region.

PREVENTIVE MEASURES

The toxic effects of arsenic affect people of all ages and mainly those living in poverty and undernourishment. Trapped in a vicious circle, the sick lose their jobs and become a burden to their families. They have no social security or medical insurance, leading them into poorer health and greater poverty.

Most of the detrimental effects of arsenic are irreversible. At an early stage, drinking good quality water and eating nutritious food rich in vitamins can reverse some of the effects.

Some suggestions to combat arsenic contamination in groundwater are:

- building up deeper wells according to a proper perforation, casing and development design; this measure depends on the improvement of the groundwater with depth.
- either using surface water from rivers or lakes, or treating collected rainfall with proper technology, so as to prevent the risk of infection. Surface water usually contains less arsenic, but at the same time, poses a greater risk of biological contamination. Due to the fact that infectious hydric diseases cause more deaths than those resulting from arsenic, it is necessary to be careful when it comes to selecting alternative sources of water supply. It should also be taken into account that river waters sometimes have a high arsenic content.
- education programmes to raise awareness about the detrimental effects that arsenic in groundwater has on health and alternative sources of water supply.
- arsenic removal systems, as well as proper final disposal sites for residual waters and solids coming from treatment plants.
- hydrogeological surveys and groundwater quality monitoring, mainly of arsenic content evolution.
- clinical monitoring for early detection of the symptoms of hydroarsenic poisoning in the population.

FINAL COMMENTS

Chronic hydroarsenic poisoning in the Republic of Argentina constitutes an important medical, sanitary and social problem. The search for new arsenic areas requires the coordinated work of hydrogeologists, chemists, economists, social workers and sanitation doctors. It is necessary to promote joint action between health ministries and water managers in order to make health an important issue in water administration.

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CHAPTER 7

Influence of irrigation on groundwater nitrate concentrations in areas considered to have low vulnerability to contamination

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ABSTRACT: In Campina de Faro, an area in the south of Portugal, agricultural practices have caused the rise of nitrate concentrations in groundwater to extremely high levels, frequently exceeding 250 mg/l in the upper aquifers. While the origin of the nitrate contamination lies in the excessive use of nitrogen fertilizers, irrigation with local, nitrate-rich groundwater contributes significantly to the problem. This additional source of nitrogen for the crops is generally not taken into account in fertilization plans. Furthermore, irrigation return flow, concentrated by strong evapotranspiration, gradually increases nitrate and overall salinity levels in the upper aquifers. Nitrate contamination is studied with respect to both its temporal evolution and spatial distribution. Indicator-geostatistical techniques are applied, resulting in maps that indicate the probability of nitrate concentrations exceeding predetermined threshold values. They indicate that the highest contamination levels occur in aquifers with high groundwater residence times and low recharge rates, enhancing the impact of the groundwater recycling process induced by irrigation. Curiously, the same hydrological conditions are generally considered by vulnerability assessment methods to decrease an area's pollution potential as they prevent rapid propagation of the contaminant. These apparent contradictions are demonstrated and analysed with the application of the DRASTIC vulnerability assessment methodology.

INTRODUCTION

Groundwater vulnerability assessment methods were developed with the idea to rapidly identify the pollution potential in large areas. Results, frequently presented in maps, can be used for regulatory, managerial and decision-making purposes. The term 'vulnerability' is itself quite ambiguous, as discussed by Vrba & Zoporozec (1994) in their

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Guidebook on Mapping Groundwater Vulnerability. The authors conclude that groundwater vulnerability can be defined correctly as an intrinsic property of a groundwater system that depends on the sensitivity of that system to human and/or natural impacts. Furthermore, they distinguish intrinsic (or natural) vulnerability from specific (or integrated) vulnerability, the first term defined solely as a function of hydrogeological factors and the latter term defined by the potential impacts of specific land uses and contaminants.

Most groundwater vulnerability assessment methods concentrate on the intrinsic component, assuming therefore that their application is valid for all potential contaminants and independent of the way they are applied. This is an important limitation of such methods, since the behaviour of different contaminants is extremely variable and complex. For example, groundwater contamination by heavy metals can be strongly retarded by the presence of clay layers (cation exchange), whereas the behaviour of nitrates, once they leave the vadose zone, can be quite conservative.

Another issue is the generalization of the influence of intrinsic factors on the pollution potential. As processes involved in groundwater contamination vary widely from case to case, it is hard to believe that the role of soil and aquifer properties is always the same. Consequently, while in many cases groundwater vulnerability to contamination is correctly estimated, in others the results may be in total discrepancy with reality. Once the maps are in the hands of the decision makers, erroneous conclusions as to which areas require more protection against contaminating activities can obviously lead to serious problems.

A clear example of such misinterpretation is given in the present case study of the effect of irrigation on groundwater nitrate concentrations in Campina de Faro. This area in the south of Portugal is characterized by intense agricultural activity and the contamination problem has been discussed by several authors (Almeida & Silva, 1987; Silva, 1988; Stigter et al., 1998; Stigter & Carvalho Dill, 2001b). The spatial distribution of nitrates is far from uniform and seems to be influenced by certain intrinsic factors. The main objective of this paper is to test whether or not these influences can be predicted by vulnerability assessment methods. One of these methods, named DRASTIC (Aller et al., 1987), is applied to the study area and the results are compared with present-day contamination levels. The spatial distribution and temporal evolution of these levels are analysed with the aid of indicator-geostatistical tools and are presented in a set of maps. These maps indicate the possibility that nitrate concentrations found in groundwater exceed predetermined threshold values. The use of this probabilistic methodology, already introduced in earlier Portuguese case studies (Ribeiro, 1998), constitutes a useful tool for groundwater management.

STUDY AREA

Location and climate

The study area of Campina de Faro is located directly north of Faro, the capital of the southernmost province of Portugal, the Algarve (Figure 1). It measures approximately 4×6 km and is bordered by the Atlantic Ocean in the south, the River Ribeira de Colmeal in the west and the River Rio Seco in the east. The northern boundary corresponds to the end of the area with intensive agricultural activity. Elevations increase

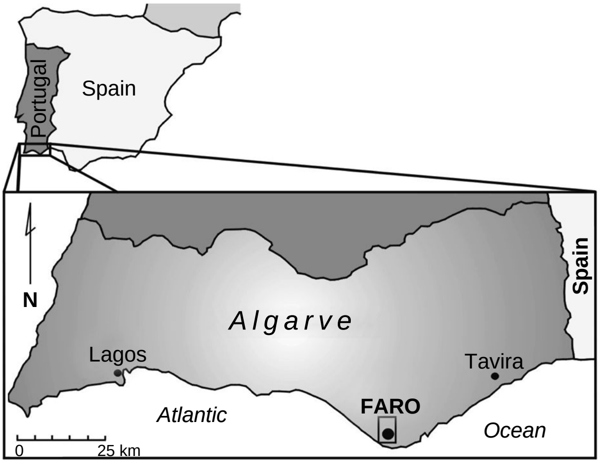

Figure 1. Location of the study area.

from 2 m above sea level in the south to 40 m above sea level in the north. The topography is generally rather flat, an exception being the hill on which the city of Faro has been built, which is associated with diapiric activity (Silva et al., 1986).

The area has a warm Mediterranean climate, with mean annual values of air temperature and precipitation, measured at Faro airport, of 17.3°C (Silva, 1988) and 531 mm (Loureiro & Coutinho, 1995), respectively. Potential evapotranspiration amounts to an average of 876 mm/year (Faria et al., 1981), thus largely exceeding precipitation. Real evapotranspiration losses vary between 70 and 85 per cent of precipitation, depending on the method of calculation (Silva, 1988; Stigter et al., 1998).

Hydrogeology

A detailed description of the geology of the area and surrounding region can be found in the work of Silva (1988). This section briefly discusses the area's hydrogeological setting, characterized by the hydrogeological map displayed in Figure 2. Three aquifers are apparent. The first aquifer system is formed by Cretaceous limestone layers separated by marls. This formation crops out in the north, dips to the south at angles of around 20–30° and is found at depths below 200 m near the city of Faro. According to Manuppella (1992), its maximum thickness exceeds 1000 m and is found just east of the study area.

Miocene fossil-rich limestones have been deposited in a graben-like structure bordered by large north-south trending faults (Silva, 1988). These sediments form the second aquifer, its thickness increasing from north to south and exceeding 200 m near the coast. Due to karstification of the limestones, aquifer transmissivity is high. No outcrops exist of this formation, since it is covered by fine sands of Miocene age, deposited in the same graben-like structure.

The Miocene sands, together with the overlying sands and gravels of Pliocene to Quaternary age, build up the uppermost, phreatic aquifer. This aquifer has a maximum thickness of about 50 m in the centre and south of the study area and has low discharge

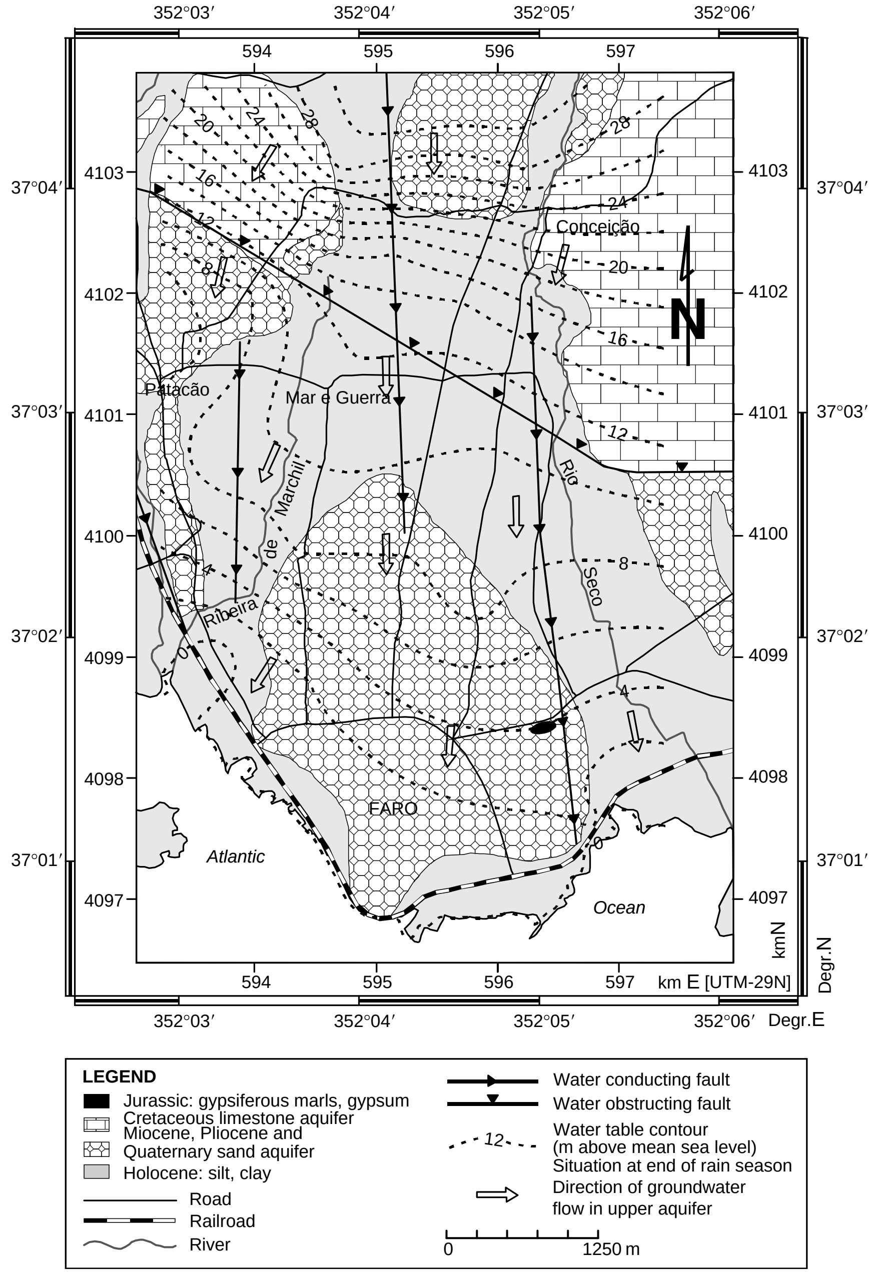

Figure 2. Hydrogeological map of the study area.

rates (Stigter & Carvalho Dill, 2000b). Fluvial and marine erosion took place during Holocene times, followed by the deposition of a thin layer of clays and silts in a large part of the area (Figure 2). However, their thickness is often too small to give the upper aquifer a confined character.

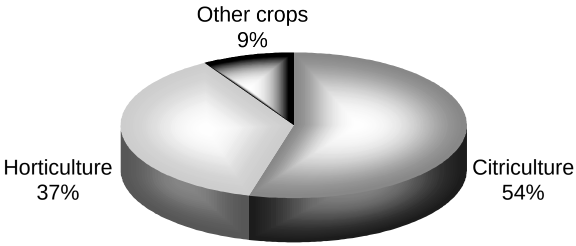

Figure 3. Current distribution of agriculture in the study area.

As indicated in Figure 2, the general direction of groundwater flow is north-south. Infiltration of rainwater takes place in the Jurassic limestones north of the Campina de Faro (not on the map) and provides abundant water for irrigation in cultivated areas near the coast. Preferential flow paths for groundwater are formed by the north-south trending faults. In contrast, the steeper gradient in the water table contours in the north seems to indicate an obstruction to groundwater flow caused by the northwest-southeast trending fault that acts as a barrier (Stigter et al., 1998).

Land use

Agriculture has been the area's main land use over centuries. Initially, the landscape was dominated by almond, fig, olive and carob trees and vineyards. These cultures had a low water demand and hence were suitable for the warm and dry climate of the area. At the end of the nineteenth century, agricultural activity entered a new phase with the construction of over 200 shallow hand-dug wells between 1889 and 1933, triggering the development of irrigation techniques (Simões, 1935). According to the same author, by the end of that period almost 40 per cent of the agricultural land was occupied by irrigated crops, including vegetables (beans, potatoes, maize) and orange trees.

At the end of the 1970s, almost all agricultural land in Campina de Faro was irrigated by local groundwater (Barradas et al., 1979). Citriculture had become the dominant land use, though horticulture was also being practised on a large scale, both in greenhouses and in the open air. The present-day situation shows a similar distribution, as indicated in Figure 3, which was derived from inquiries held among the farmers of the study area.

Greenhouse agriculture requires the application of large amounts of water, fertilizers and pesticides to allow intensive cultivation. However, regarding the application of nitrogen (N), citrus culture is just as demanding. An amount of 150–300 kg/ha yr of N is recommended for citrus trees, whereas for greenhouse crops such as tomatoes and melons (both cultivated in the study area), quantities are considered per crop cycle (± half a year) and equal 150–200 kg/ha and 50–100 kg/ha, respectively (Quelhas dos Santos, 1991).

As for water demand, Shalhevet and Levy (1990) indicate values of 500–1,000 mm/yr for citrus yards cultivated in Israel. Stanley and Maynard (1990) point to values of 400–600 mm per crop cycle for tomatoes and melons. A mean value of 1,000 mm/yr or 10,000 m^3/ha yr of groundwater extracted for irrigation were the values derived from inquiries held among the farmers in the region and used by Stigter et al. (1998) in water balance calculations.

PROBABILITY MAPS

Methodology

Probability maps are produced by a set of robust geostatistical tools, involving variographical analysis and kriging of indicator variables. In this non-parametric approach (not restricted to any Gaussian distribution), the probability distribution function of the unknown value in any location, conditional to the available information, is analysed (Journel, 1987; Isaaks & Srivastava, 1989).

Figure 4 provides an overview of the steps that need to be carried out in order to create probability maps. In a first phase, the original data are converted to indicator values on the basis of specific cut-off levels, which define the indicator variables. These are characteristic values defined either by drinking water guidelines or data percentiles (such as the median). Transformation is usually binary, applying 0 to all values below or equal to the cut-off level and 1 to the remaining values.

The following step involves the structural analysis of the spatial distribution of the indicator variables, represented by their variogram. The theoretical variogram ($2\gamma(h)$) is the expected squared difference in value between pairs of data a distance and direction $h(\pm\Delta h)$ apart. On the other hand, the data's experimental variogram ($2\gamma^*(h)$) is calculated as the arithmetic mean of these squared differences (Journel, 1987; Clark, 1979). Values of $\gamma^*(h)$ are plotted against h in order to obtain the experimental function of the semi-variogram. Different spatial orientations should be analysed to discover the presence of anisotropy.

The analysis of a variable's spatial structure through its semi-variogram reveals whether or not the data become independent of one another at a certain distance. If they do, this distance and the associated semi-variogram value are defined as the range of influence and sill, respectively. Small-scale variability and measurement errors account for a certain fraction of the total data variance, known as the nugget effect and visible at the semi-variogram's origin.

In the following stage, the variable's spatial distribution is modelled by adjusting a theoretical model to the experimental semi-variogram. The obtained parameter values (anisotropy, range, sill, nugget effect) are then introduced into a kriging algorithm to estimate values at any unknown location, through a linear combination of the values at sampled locations (based on their 'structural' rather than their 'Euclidean' distances).

In the specific case of indicator variables and their binary-coded data, the resulting interpolated values vary between 0 and 1. When multiplied by 100 (%), they represent the probability that the actual value exceeds the threshold value.

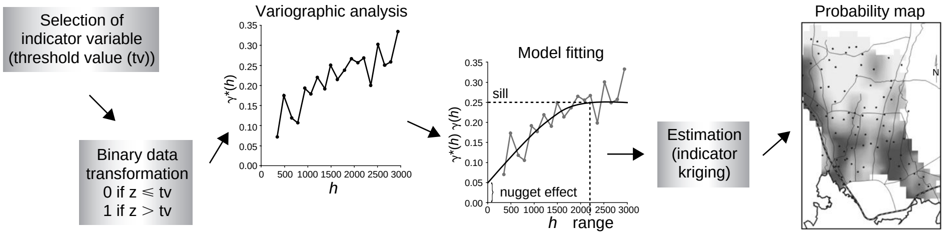

Figure 4. Schematic representation of the geostatistical procedures that results in probability maps.

Application to the study area

Groundwater contamination by nitrates in Campina de Faro was studied with respect to its temporal evolution as well as its present-day spatial distribution. For this purpose it was necessary to gather both old and recent data regarding nitrate concentrations in the area's groundwater. Recent data were collected in field campaigns between 1996 and 1998 and analysed at the laboratory of the Vrije Universiteit, Amsterdam in the Netherlands (Stigter et al., 1998; Stigter & Carvalho Dill, 2001b). With regard to the earlier period, the interval 1978–81 was chosen, due to the availability of data kindly provided by the Technische Universität, Berlin in Germany (Dominik et al., 1980) and the DRAOT-Algarve, the Regional Directorate for the Environment and Territorial Planning of the Algarve in Portugal. Data collection and optimization sought to produce a spatial sample distribution as uniform as possible. The geostatistical treatment only considered data of the upper aquifers, with maximum sampling depths of 60 m, corresponding more or less to the depth of the upper aquifer in the centre and south of the study area. Including the deeper aquifer systems, with much lower nitrate concentrations, would only disturb the overall image in a forced attempt to present both horizontal and vertical variations. Table 1 briefly characterizes the nitrate data of both periods. Concentrations are expressed in mg NO_3^-/l throughout this chapter.

In a next stage the indicator variables for which the probability maps would be produced were selected. The maximum admissible nitrate concentration in drinking water was chosen as a cut-off value for both periods with the aim of studying temporal evolution. The present-day spatial configuration of the contamination plume was examined by creating probability maps of the cut-off values associated with the data's 1st quartile, median and 3rd quartile. All experimental variograms were analysed with respect to their spatial structure and orientation, and theoretical spherical models were adjusted. Table 2 lists the model parameter values and Figure 5 illustrates the experimental variograms and their adjusted models.

Results and discussion

When observing the variograms of the indicator variable 50 mg NO_3^-/l for both periods, it becomes clear that the variable is better structured for the earlier period, in a north-south direction, possibly related to the direction of groundwater flow in the upper aquifer systems (Figure 2). The nugget effect only accounts for a minor part of the total variance. This is not the case for the recent period, which shows little variance, a relatively high nugget effect and no preferential structural orientation. With respect to the variograms of the indicator variables defined by the recent data's quartiles, the best behaviour is shown by the data's median, with its corresponding threshold value of 161 mg NO_3^-/l. Both this

Table 1. Nitrate data characteristics.

Period	Nr. of samples	NO_3^- (mg NO_3^-/l)			
		Minimum	Median	Mean	Maximum
78–81	81	0	48	74	254
96–98	76	30	161	189	581

Table 2. Main orientation of experimental variograms and parameter values of adjusted spherical models.

Period	Indicator variable	Orientation of variogram	Spherical model		
			c_0	c	A
78–81	50 mg/l (MAC)	90°	0.050	0.200	2200
96–98	50 mg/l (MAC)	Omni	0.040	0.074	2100
	1st quartile (88 mg/l)	90°	0.075	0.113	1200
	median (161 mg/l)	90°	0.080	0.220	2300
	3rd quartile (253 mg/l)	Omni	0.055	0.165	2400

c_0 = nugget; c = sill; A = range; Omni = omnidirectional, 0°–180°
MAC = maximum admissible concentration for drinking water.

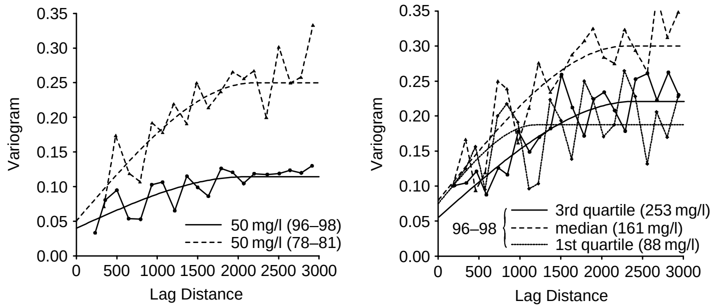

Figure 5. Experimental variograms of the data and adjusted spherical models.

variogram and the one of the 1st quartile are best structured in a north-south direction, once again referring to a possible connection with groundwater flow.

Figure 6 displays the probability maps for the indicator variable 50 mg NO_3^-/l for both periods. What these maps actually indicate is the chance that nitrate concentrations in groundwater sampled in the upper aquifers exceed the maximum admissible concentration in drinking water (50 mg/l). The observed nitrate data form the basis for these estimations.

The situation at the end of the 1970s and beginning of the 1980s clearly points out the existence of several areas where nitrate concentrations in groundwater of the upper aquifers were likely to exceed the drinking water limit. As Barradas et al. (1979) point out, agriculture at the time was already almost entirely focused on irrigation supplied by local groundwater. The higher contamination levels in the centre and south could be a result either of more intense agricultural activity or the specific hydrogeological characteristics of this area, or a combination of the two.

No detailed information could be gathered with respect to the first factor, but the last factor is of extreme importance, as discussed by Stigter and Carvalho Dill (2001a). The authors carefully describe the groundwater cycle induced by irrigation with local groundwater, which is schematized in Figure 7. The irrigation return flow is highly concentrated, especially due to strong evapotranspiration (mainly water uptake by crops, but also some evaporation). Although initially attenuated through mixing with resident groundwater, overall salinity and nitrate concentrations in the aquifer gradually increase

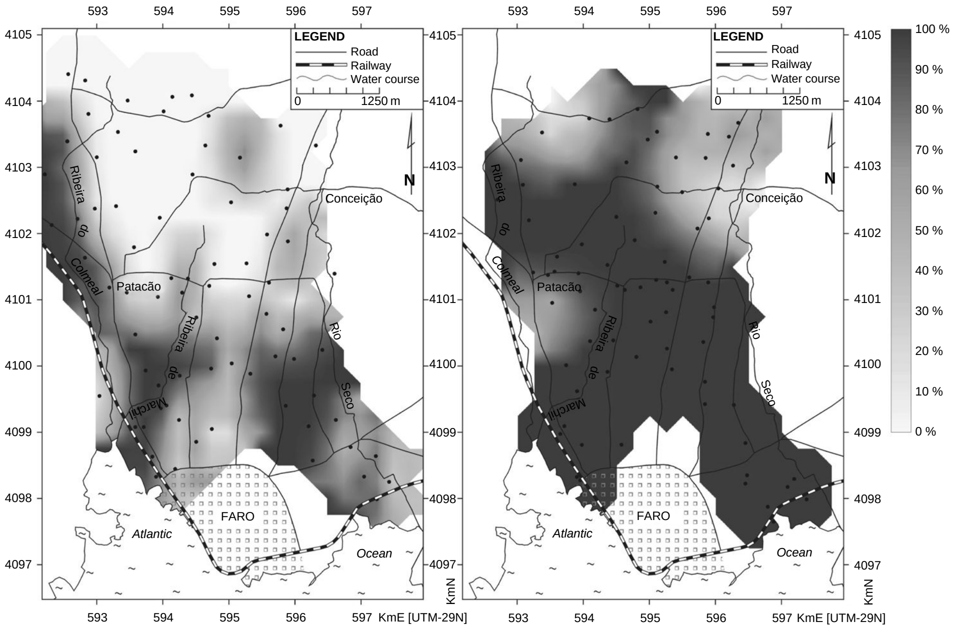

Figure 6. Probability maps of the indicator variable 50 mg NO_3^-/l for earlier period (left) and recent period (right).

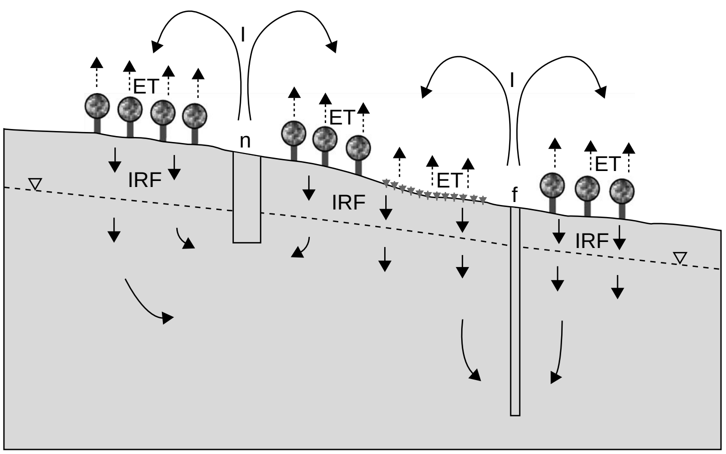

Figure 7. Groundwater cycle induced by extraction from local wells for irrigation (I); ET = evapotranspiration, IRF = irrigation return flow, $\underline{\nabla}$ = water table.

as the groundwater recycling process (Stigter & Carvalho Dill, 2001a) repeats itself. Low aquifer recharge rates and high groundwater residence times benefit the efficiency of the process, since these characteristics prevent the groundwater from leaving the system rapidly and thus promote its recycling. This is precisely what occurs in the upper sand aquifer present in the centre and south of the study area. However, in the north, the upper aquifer system consists of Cretaceous limestone layers and although separated by marls, they are highly karstified and show a rapid water table response. As a consequence, groundwater residence times are low and conditions for continuous groundwater recycling do not exist.

As Table 1 indicates, the median value of the earlier nitrate data is slightly below the drinking water limit. The average concentration is somewhat higher, indicating the presence of a few extreme values (such as the maximum value of 254.2 mg/l). However, these constituted local anomalies and on the whole the application of nitrogen fertilizers certainly was not as intensive at that time as it would be in a later stage (more towards the middle and end of the 1980s). Agricultural practices continued to develop and intensify with the drilling of new wells and the massive use of chemical fertilizers, consequently increasing the load of contaminants on groundwater quality. Figure 6 reveals that nowadays the possibility of nitrate concentrations in groundwater exceeding the drinking water limit is 90–100 per cent in almost the entire study area (this explains the low sill in the variable's variogram).

It is interesting to study the present-day shape of the contamination plume in order to find out if the groundwater recycling process continued to have its effect on nitrate contamination. If so, the highest nitrate levels should still be found in the sandy aquifer in the centre and south of the study area. Probability maps of the indicator variables defined by the data's 1st, 2nd (median) and 3rd quartile values were created and are presented in Figure 8.

A first observation can be made on the data's 1st quartile value of 88 mg NO_3^-/l, meaning that 75 per cent of the data largely exceed the maximum admissible concentration in drinking water. This is a worrying fact since a large share of the area's rural population is forced to use local groundwater for household purposes due to the lack of a public water supply (although groundwater of higher quality can be extracted from deeper wells).

The probability maps of the first quartile and, to a lesser extent, the median, show a north-south orientation of the high-risk areas. In the north these areas are not so extended, but where they exist, they connect with the south, apparently related to the north-south orientation of the groundwater flow. The north-south connection already fades slightly on the map indicating probabilities of groundwater nitrate values exceeding the median value. Finally, in the probability map of the 3rd quartile, links between north and south have totally disappeared. In other words, high probabilities of nitrate concentrations exceeding 253 mg/l (five times the maximum admissible concentration in drinking water) are restricted to the centre of the study area.

The converging of the high-risk area around the centre of the study area with increasing percentiles clearly defines the shape and location of the contamination plume. Its location corresponds to the presence of the upper sand aquifer, where the groundwater recycling process is indeed most effective, as was expected from its high residence time. The presence of the northwest-southeast trending fault just north of this area (Figure 2) could also be an influencing factor due to its behaviour as a flow barrier.

The Cl^- ion, considered to have a conservative behaviour in hydrochemical terms, can be used as an indicator of the groundwater recycling process, since its mass balance reflects dilution, concentration and mixing processes. It was already shown by Stigter et al. (1998) that there is a significant increase of the Cl^- concentration in the upper sand aquifer of Campina de Faro.

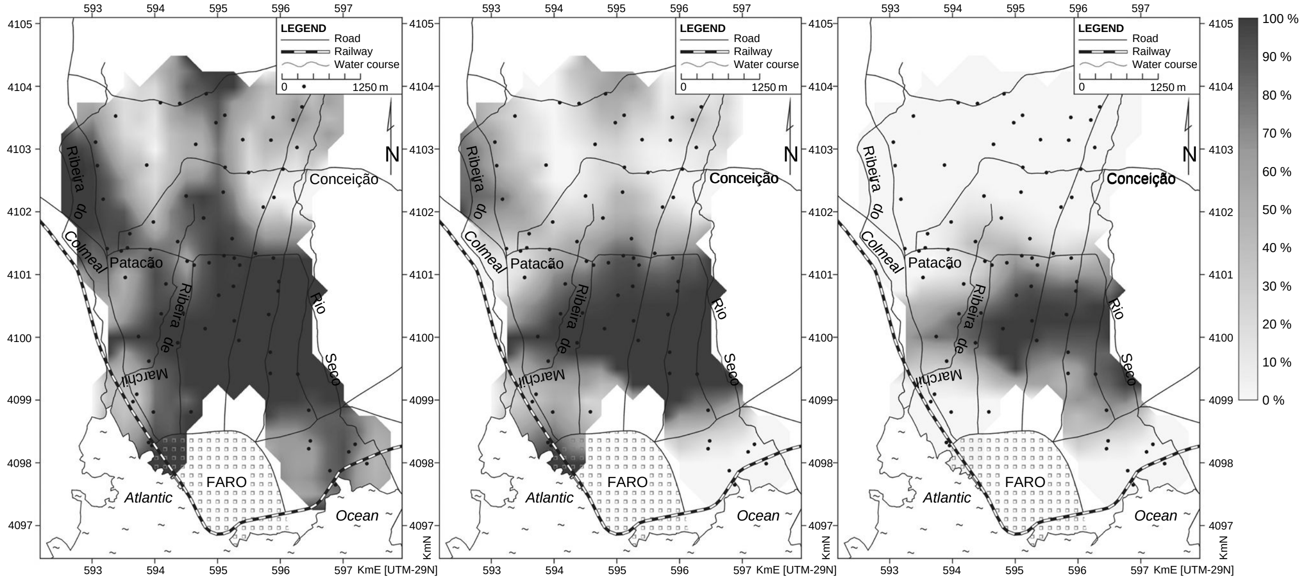

Figure 8. Probability maps of indicator variables associated with the recent data's 1st quartile, median and 3rd quartile.

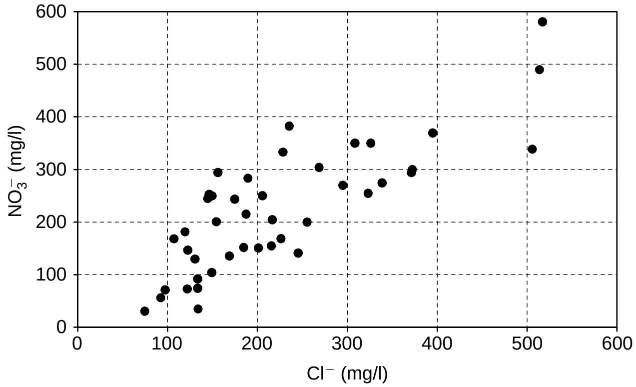

Figure 9. Plot of NO_3^- versus Cl^- for groundwater samples in study area.

An interesting fact is that the NO_3^- concentration follows the same trend. Figure 9 is a plot of Cl^- versus NO_3^- concentrations of nearly all the (shallow and deep) groundwater samples gathered in the study area between 1996 and 1999. Only samples of groundwater that suffered seawater intrusion were left out off the graph, since the additional Cl^- source biases the $NO_3^-{-}Cl^-$ relationship. The plot clearly indicates that the Cl^- concentration increase induced by the groundwater recycling process is accompanied by an increase in NO_3^- concentrations. This suggests a remarkably conservative behaviour of NO_3^-, in spite of its concentration being influenced by many additional factors, such as fertilization, uptake by crops and complex biogeochemical processes in the soil and unsaturated zone. Whereas the latter processes possibly explain the scatter around the linear relationship between NO_3^- and Cl^-, it should be concluded that the irrigation practices have a strong influence on the increase of both NO_3^- and Cl^-.

High nitrate concentrations in irrigation water should be taken into account when calculating nitrogen demands of crops and establishing a fertilization plan. Unfortunately, such efforts are rarely reported, particularly because of the lack of information and technical support offered to the farmers of the region.

VULNERABILITY MAPS

DRASTIC methodology

The intrinsic vulnerability assessment method, DRASTIC was developed by Aller et al. (1987) at the US Environmental Protection Agency (EPA). The idea of the method was to systematically evaluate the pollution potential of any hydrogeological setting throughout the US (Aller et al., 1987). The seven letters that make up the acronym DRASTIC represent seven hydrological parameters incorporated into a relative ranking scheme that uses a combination of weights and ratings to produce a numerical value called the DRASTIC index (Table 3).

For a detailed description of the methodology and its application, the reader is referred to the work of Aller et al. (1987). The DRASTIC index values range from 23 to 226 and are distributed among eight classes when displayed on a map. The higher the index, the

Table 3. Parameters considered by the DRASTIC index and assigned weights (after Aller et al., 1987).

Letter	Parameter	Weight
D	Depth to water	5
R	Net recharge	4
A	Aquifer media	3
S	Soil media	2
T	Topography	1
I	Impact of the vadose zone media	5
C	Hydraulic conductivity of the aquifer	3

greater the vulnerability to groundwater contamination. The following conditions all contribute to a higher DRASTIC vulnerability index:

- low depth to groundwater
- high net aquifer recharge
- permeable aquifer media showing no reactivity with regard to the contaminant
- soil media lacking clay and organic material
- flat topography
- permeable vadose zone media showing no reactivity with regard to the contaminant
- high hydraulic conductivity of the aquifer.

Application to the study area

The DRASTIC methodology was first applied in Portuguese case studies by the National Laboratory of Civil Engineering (Laboratório Nacional de Engenharia Civil; LNEC). Applied both on a national and a more regional scale, one of those applications involved the evaluation of the vulnerability to pollution of aquifer formations of the coastal areas (Lobo-Ferreira et al., 1995), including the study area, on a scale of 1:100,000.

Due to the availability of more and recent data, it was thought useful to calculate the DRASTIC index for the study area once more on a smaller scale, namely 1:25,000. New data on aquifer material, groundwater levels and aquifer recharge (including irrigation, to be considered as an additional input, according to Aller et al. (1987)) had been gathered over the period 1996–99, which proved useful for the parameter calculations. Table 4 indicates the data sources used to classify each of the seven parameters. Parameter rating was based on the tables supplied by Aller et al. (1987) and, in case of doubt, the work of Lobo-Ferreira et al. (1995) was consulted.

A regular grid with a spacing of 250 m was laid over the study area and for each grid point the parameter values were determined and the final index was calculated as the weighed sum of those parameters. The vulnerability map was then composed by drawing a contour plot based on the final grid point values, with the contour intervals linked to the eight vulnerability classes defined by Aller et al. (1987).

Results and discussion

The DRASTIC vulnerability map obtained of the study area is displayed in Figure 10. The centre and south of the study area are considered by DRASTIC to have moderate

Table 4. Data sources of DRASTIC parameters.

Parameter	Data source
D	Monthly monitoring of 32 shallow wells between 1997 and 1999, resulting in water table time series
R	Data on precipitation (Loureiro & Coutinho, 1995), irrigation (inquiries held among farmers; Shalhevet & Levy, 1990; Stanley & Maynard, 1990) and evapotranspiration (Silva, 1988; Stigter et al., 1998)
A	Geological information (Silva, 1988; Stigter et al., 1998) and analysis of groundwater level time series
S	Characteristics of soils in the Algarve (Kopp et al., 1989)
T	Percentage slope values determined from elevation contour lines on topographical maps of scale 1:25,000
I	Geological information (Silva, 1988; Stigter et al., 1998) and analysis of groundwater level time series
C	Geological information (Silva, 1988; Stigter et al., 1998)

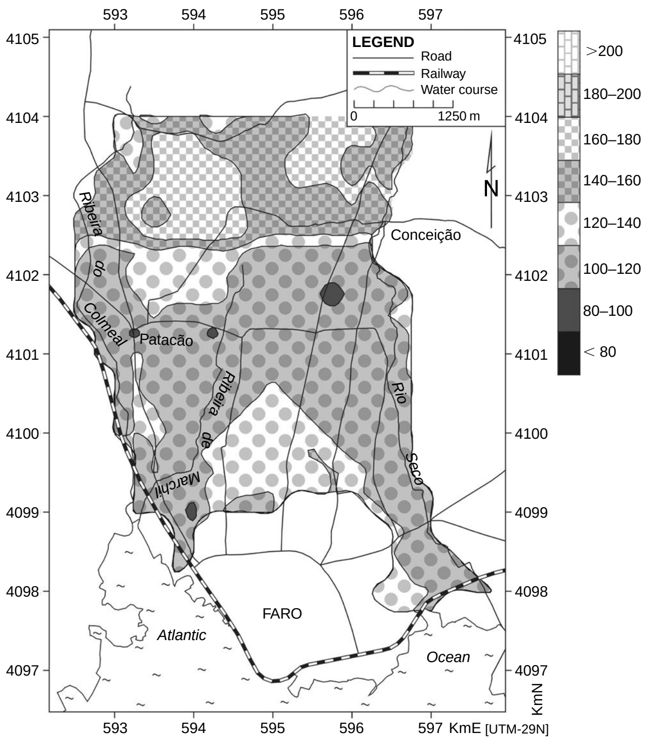

Figure 10. DRASTIC vulnerability map of the study area.

vulnerability to groundwater contamination. Although recharge is high (due to irrigation) and the water table is found at relatively shallow depths, the moderate score of the DRASTIC index is caused by the low hydraulic conductivity of the upper sand aquifer and the derivation of soil and vadose zone from Holocene clays and silts.

In contrast, the presence of Cretaceous karstic limestone layers in the north, with high hydraulic conductivities (in spite of being separated by marls), high net recharge and low depth to water table, results in a high index calculated by DRASTIC, hence indicating high vulnerability to groundwater contamination.

The vulnerability map shows some discrepancies with the probability maps discussed in an earlier section. In fact, it appears that the least vulnerable areas coincide with those where chances of detecting extreme levels of nitrate contamination are highest. This is largely due to the fact that the hydrogeological conditions that lower the DRASTIC index in Campina de Faro are precisely those that strongly promote groundwater recycling, which proved to be a crucial factor on the degree of contamination.

CONCLUSIONS

Although the excessive application of nitrogen fertilizers is undoubtedly the trigger of groundwater contamination by nitrates, it is not the only controlling factor on the degree of contamination in the study area. Instead, irrigation with locally extracted groundwater induces a groundwater cycle that gradually increases the nitrate concentrations along with the overall groundwater salinity. High groundwater residence times increase the efficiency of the process, which explains the higher level of contamination in the upper sand aquifer of the study area.

The probability maps, created with the aid of an indicator-based geostatistical methodology, constitute a useful tool in the spatial structural analysis of the contamination plume and the related uncertainties. Furthermore, they allow a quick comprehension of the state of contamination with respect to a certain health standard, such as the maximum admissible concentration in drinking water. However, as probability maps are based on real observations, an irregular spatial distribution or simply a lack of data can lead to maps of much lower significance.

Regarding the drinking water limit for nitrates (50 mg/l), it can be concluded that at the end of the 1970s, high probabilities of nitrate concentrations exceeding this value in shallow groundwater (up to 50 m depth) already existed in considerably large areas, especially in the centre and south. Nowadays, these probabilities amount to almost 100 per cent in the entire study area.

As for the DRASTIC method, it does not succeed very well in the vulnerability assessment of the study area. Although an area's intrinsic vulnerability does not necessarily correspond to its pollution risk (the latter also depending on the presence of a contamination source), there is a clear reason for the observed discrepancies. The hydrogeological conditions believed by DRASTIC to involve a high vulnerability to groundwater contamination, indeed promote a rapid arrival of the contaminant to groundwater, as well as its spreading through the aquifer. However, they equally contribute to the contaminant's dilution and attenuation. On the contrary, low recharge rates and permeabilities of aquifer and vadose zone, supposedly conditions of low vulnerability, favour the persistence and accumulation of the contaminant. In the present case study, this effect is enhanced by the occurrence of the groundwater recycling process. These factors are not accounted for by vulnerability assessment methods such as DRASTIC, which explains why their results do not always correspond to reality.

ACKNOWLEDGEMENTS

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CHAPTER 8

Nitrogen impacts from a septic system in an unconfined aquifer in São Paulo, Brazil

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ABSTRACT: Groundwater impacts caused by septic system effluents in an unconfined sandy alluvial aquifer at Parque Ecológico do Tietê (São Paulo, Brazil) were investigated through the chemical analysis of 68 shallow wells in a 50×50 m area. Chemical parameters, such as nitrate, nitrite, ammonium and chloride, were measured biweekly on site, and major ions and other nitrogen compounds were analysed monthly in the laboratory. This monitoring programme identified two different hydrochemical environments: calcium-sulphate chloride water and sodium-nitrate chloride water. Due to the distribution of nitrogen species within the sodium-nitrate chloride plume, it was possible to identify three distinct geochemical zones: (i) organic nitrogen + ammonium, (ii) ammonium + nitrate and (iii) nitrate. Near the septic tanks, there was a predominance of reduced compounds, due to oxygen consumption by organic matter degradation and nitrification processes that promote a decrease of pH values. Concentrations of nitrogen compounds, and other chemical parameters, change during dry and humid seasons as a consequence of recharge, which in turn cause changes in the geometry of streamtubes. Groundwater flow and plume extension are highly dependent upon such aquifer properties as recharge and spatial variations in hydraulic conductivity in the area.

INTRODUCTION

Nitrate is the pollutant most frequently found in groundwater. At concentrations exceeding 10 mg/l NO_3^--N (approximately 44 mg/l NO_3^-) it can cause methemoglobinemia and cancer (WHO, 1999; USEPA, 1995). Nevertheless, EC Directive (91/676/EEC) proposed that concentrations of this compound could reach values of up to 11.3 mg/l NO_3^--N (equivalent to 50 mg/l NO_3^-). Agricultural practices, including the use of organic and inorganic fertilizers, cultivation of virgin areas and cattle raising are important anthropic sources of this particular contaminant. Another source is associated with on-site sanitation systems, either in the form of septic tanks or cesspits. In areas without main drainage, which are not served by sewers, inadequate construction of septic tanks, as well as the lack of maintenance of these systems, can contribute to the contamination of shallow aquifers, especially in areas of high population density (Foster & Hirata, 1988).

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According to IBGE (1991), 42 per cent of the Brazilian population (approximately 61 million people) either use cesspits or do not possess any kind of sanitation system whatsoever. Consequently, there is inadequate disposal of liquid effluents, even directly into the aquifer (where the base of a cesspit reaches the water table). The presence of 'favelas' (shantytowns) aggravates this situation due to high population density and the high concentration of cesspits sited very close to hand-dug wells (Ferreira & Hirata, 1993). The proximity of cesspits to water wells is not restricted to shantytown areas. According to the same census, the water supply for 20 per cent of Brazil's population comes from hand-dug wells and only 32 per cent (around 45 million people) is connected to public mains drainage.

Studies of nutrient behaviour and attenuation date back several years (Keeney, 1986; Robertson et al., 1991; Wilhelm et al., 1994; Hirata et al., 1997). Apart from this, specific studies on the contamination of aquifers by nitrogen, including detailed monitoring programmes, are rare in Brazil, and Latin America in general.

This study was conducted with the objective of assessing the impacts of an on-site sanitation system on an unconfined aquifer. Its principal aim was the detailed characterization of the geochemical behaviour and evolution of the nitrogenated series in the contamination plume, by means of 68 monitoring wells, installed in an area of 2,500 m^2.

STUDY AREA

The study area is located in the 'Parque Ecológico do Tietê Engenheiro Goulart – PET-EG' (Tietê Ecological Park, Engenheiro Goulart Leisure Centre) in the eastern part of the municipality of São Paulo. The PET-EG was created in 1976 with the aim of preserving the Tietê alluvial flats, as well as reducing the impact of floods in this part of Metropolitan São Paulo (Figure 1).

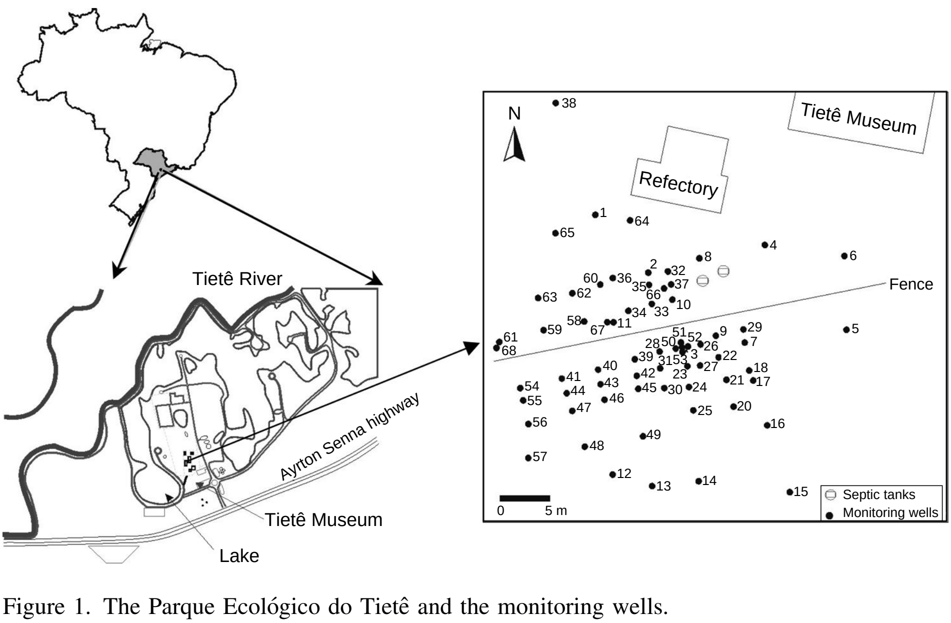


Figure 1. The Parque Ecológico do Tietê and the monitoring wells.

The Tietê Museum, formerly known as the 'Casarão', was the residence of the owner of the rural property that existed in the area before the inauguration of the PET-EG. This building now serves as a reception area for students taking part in environmental education programmes. Normally, approximately 60 young students visit the museum every day, making up a total of 22,000 people per year. The two-storey building has four bathrooms. Close to the 'Casarão' there is a workers' refectory, which has two bathrooms.

All the sewage from the two buildings drains to a septic system 30 m to the south of the museum. This system is made up of two concrete tanks with diameters of 1.5 m and depths of 1.0 m (each having a volume of 72 m^3), which have never been cleaned since the creation of the PET-EG.

The first tank receives raw effluent, which is transferred to the second tank by means of a connection in the upper part of its structure. The second tank allows the liquid to infiltrate into the soil through openings in its side walls. The bases of the two tanks are closed.

Monitoring well installation

A total of 68 monitoring wells were installed in an area of 2,500 m^2. Of this total, 65 wells were installed to a depth of 3.0 m and three to a depth of 5.0 m (Figure 1). The 3-m wells were drilled using a hand auger with a diameter of 0.1 m. Piezometers, made of PVC tubes with a diameter of 0.01 m and the last 0.5 m perforated and covered with a bidim screen (filter section), were installed in the area. A sand pack and bentonite seal were used to fill the annular space.

The three wells (wells PET66, PET67 and PET68) with a depth of 5.0 m were drilled using a Piönjar portable gasoline-powered pneumatic hammer with a 0.5 m filter section. Galvanized steel tubes were driven into the ground. Each of these three wells was drilled at a distance of approximately 1 m from wells PET37, PET11 and PET60.

Sample collection

The water was sampled by mean of a Geopump II peristaltic pump coupled to a holder, with a Millipore cellulose acetate filter (0.45 μm). Due to high concentrations of material in suspension, it was necessary to add a pre-filter to the system. The water samples were kept in polyethylene flasks at a temperature of 4°C and sent to the laboratory on the same day.

Some chemical parameters such as NO_3^-, NO_2^-, NH_4^+, Fe^{2+} and Cl^- were measured fortnightly in the field, using Rqflex-Plus (Reflectometric Method; Merck) equipment. Alkalinity was determined in the field using titration, with 0.10 N sulphuric acid and a mixed indicator. Besides these chemical species, water level, pH, Eh, electrical conductivity, water temperature and dissolved oxygen were also measured at the same frequency, using portable equipment.

Nitrogenated compounds (N_{total}, N_{org}, NH_4^+, NO_2^-, NO_3^-) and major ions (Ca^{2+}, Fe^{2+}, Fe^{3+}, Mg^{2+}, K^+, Na^+, SO_4^{2-}, Cl^-) were measured on a monthly basis. The sample preservation procedures and analytical methodologies were in compliance with the criteria adopted by APHA (1995) and USEPA (2001).

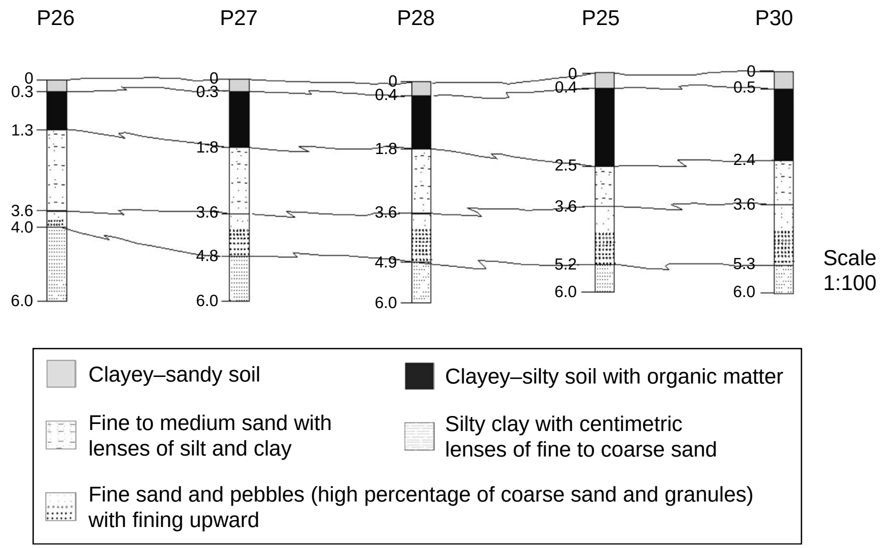


Figure 2. Geological profiles.

LOCAL HYDROGEOLOGY

The area is characterized by the presence of an unconfined aquifer of primary porosity, which consists of Quaternary Tietê River alluvial sediments with an average thickness of 20 m, as confirmed by well-logging data in the area (Figure 2). The horizontal hydraulic conductivity of these sediments ranges from 1.5×10^{-7} to 6.8×10^{-5} m/s. This material is hydraulically connected with underlying Tertiary sediments that are related to the São Paulo Formation. The average thickness of these sediments is greater than 100 m, as shown by a deep groundwater well installed close to the Tietê Museum.

The 68 monitoring wells pass through the Quaternary alluvial sediments, represented by light to dark brown intercalated lenses of clay, silt and sand, with a rich organic material content in the first 0.5 m. From 2.0–8.5 m, more permeable dark brown and yellow unconsolidated fine sands were found (Figure 2).

Aquifer recharge occurs throughout the whole area where Quaternary sediments crop out, whereas discharge takes place at streams and artificial lakes to the west of the area. The recharge is much greater during the rainy months of the year (December to March). The direction of groundwater flux is controlled by the topography, artificial lakes and the infiltration of septic tanks. Monthly potentiometric maps show two main directions of groundwater flux – one to the north and the other to the northwest. The septic system, considered to be the point of greatest hydraulic head in the area, gives rise to a radial distribution of equipotential lines from its centre (Figure 3).

The equipotential contour lines show a complex distribution and the presence of 'islands' of highest and lowest hydraulic head values, which change over time. Besides the general directions of groundwater flux, with a main northerly trend, it is possible to observe small, localized changes in flux direction. This effect can be attributed to heterogeneities in recharge patterns and hydraulic conductivity, as well as the irregular distribution of trees in the area (effects of interception and evapotranspiration).

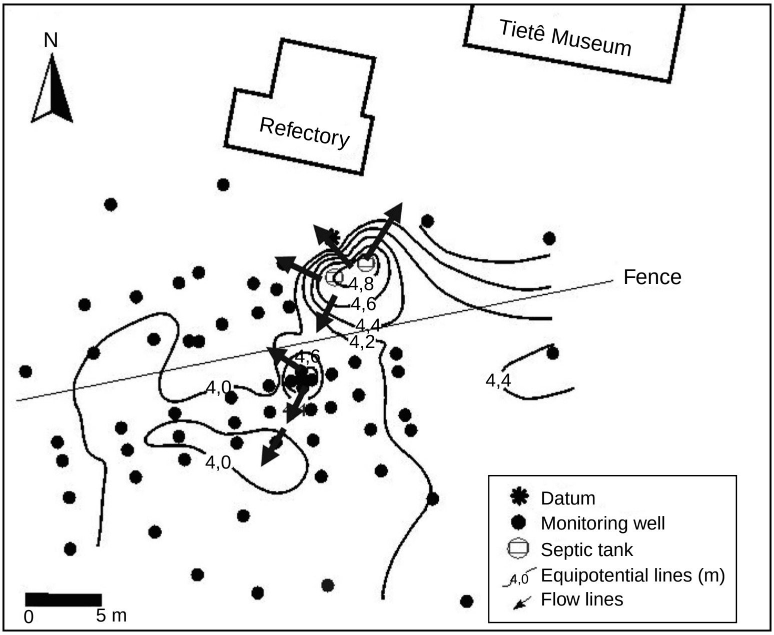


Figure 3. Potentiometric map (July/2000).

RESULTS AND DISCUSSIONS

Chemical composition of effluent in septic tanks

Very high organic, total and ammoniacal nitrogen values, with low concentrations of nitrate and nitrite were found in effluents from the two tanks of the septic system (Table 1). Ammoniacal nitrogen were present at levels of up to 80 per cent in the infiltration tank and 50 per cent in the solids tank. These concentrations were compatible with those found in the literature, where values ranged from 30 to 111 mg/l NH_4^+-N (Andreoli et al., 1979; Robertson et al., 1991; Wilhelm et al., 1996).

Precarious maintenance and inappropriate design of the septic system have caused leakage from the first tank, bringing about aquifer and soil contamination. From the point of view of sanitary engineering, the system is operating more like a cesspit than a septic tank.

Temporal variation of hydraulic parameters in the aquifer

Throughout 25 field campaigns carried out from November 1998 to April 2000, variations in the behaviour of physical-chemical parameters and water level were observed. The greatest water level depths were recorded during the dry months of the year (April to August). Recovery occurred during the months of highest rainfall (November to March). The average water level depth during the dry season was 2.16 mbs (meters below surface) and during the rainy season, 1.71 mbs.

Table 1. Effluent composition of septic tank.

Parameter (mg/l)	Solid tank	Infiltration tank
Alkalinity	537	336
Aluminium	9.72	1.64
Ammonia as nitrogen	37.0	60.0
Calcium	15.6	16.7
Chloride	93.0	105
Total iron	4.15	0.78
Phosphate as phosphorus	0.85	0.68
Magnesium	3.30	3.15
Manganese	<0.10	<0.10
Nitrate as nitrogen	0.50	1.20
Nitrite as nitrogen	0.16	<0.06
Organic nitrogen	32.0	7.30
Total nitrogen	85.0	80.0
Potassium	27.3	44.7
Sodium	63.1	24.2
Sulphate	39.0	24.0
Sulphide	<0.05	0.09

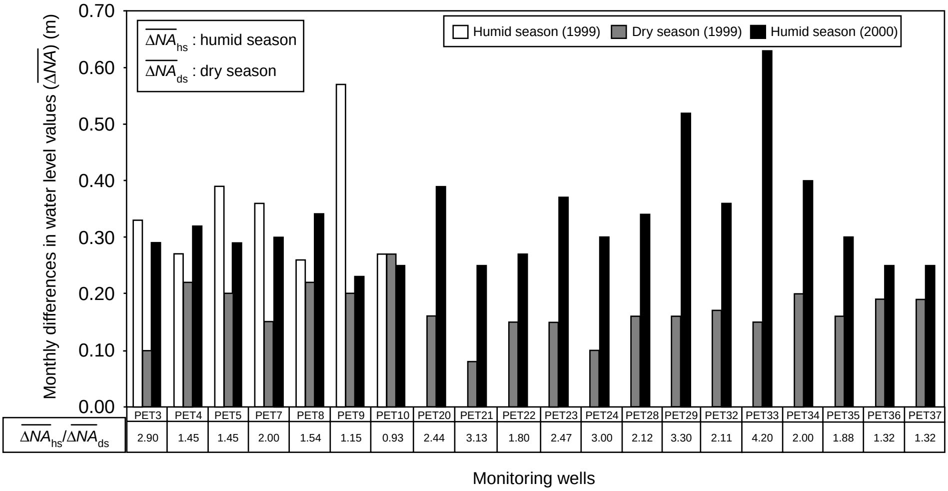


Figure 4. Differences in water levels ($\overline{\Delta NA}$) during dry and humid seasons (monthly averages).

The effects of this seasonality are depicted in Figure 4, which illustrates the average values of monthly differences in water level for each monitoring well in the area. The differences in water level between two consecutive months ($\overline{\Delta NA}$) were calculated for each well, according to the Equation 1:

$$\overline{\Delta NA} = \frac{1}{n} \times \sum_{i=1}^{n} |\mathrm{NA}_{\text{consecutive month}}\ i - \mathrm{NA}_{\text{previous month}} i\,|; n = \text{number of wells} \qquad (1)$$

During the rainy season, these differences are greater than those found during the dry season. Another factor that should be highlighted is that these differences are not evenly distributed throughout the whole area. Very high $\overline{\Delta NA}$ and $\overline{\Delta NA}_{\text{rainy}}/\overline{\Delta NA}_{\text{drought}}$ values are observed in such wells as PET33, PET29, PET20 and PET23, when compared with the remaining wells. A possible cause of this is the heterogeneity of horizontal hydraulic conductivity in the aquifer. At these four wells, higher hydraulic conductivity (K) values were observed, when compared to the average K (K^*) of the surrounding wells. This distribution of hydraulic conductivity causes heterogeneity in the dispersion of water that infiltrates from recharge. The relationship between K and K^* was established considering a radius of influence (r_i) equal to 3.5 m. This radius of influence (r_i) was divided into three sectors of equal area, forming an angle of 120 degrees. For the purposes of calculation, only those wells were considered that had at least one neighbour in each sector.

Irrespective of seasonal fluctuations, rapid variations of aquifer water level at intervals of less than a month were observed and were due to daily or accumulated weekly rainfall. In well PET9, these fluctuations were of up to 0.93 m in less than 15 days, during a rain event in the dry season, although they were more conspicuous during the rainy season. The study area favours recharge through local precipitation. These characteristics are as follows: flat area, where surface run-off is greatly reduced; soils rich in organic matter and shaded by trees, which maintain high humidity levels in the soil throughout the year; and rather shallow groundwater levels (<3.5 m).

Chemical composition of groundwater and nitrogenated series evolution in the aquifer

The 81 complete water samples, with analytical errors of less than 10 per cent, were plotted on a Piper diagram, which made it possible to distinguish two predominant hydrochemical types: sodium nitrate-chloride water (73 per cent) around the septic tanks and calcium sulphate-chloride water (27 per cent) in areas less than 10 m away from the tanks. Besides these principal types of water, two sodium bicarbonate water samples were identified close to the septic tanks.

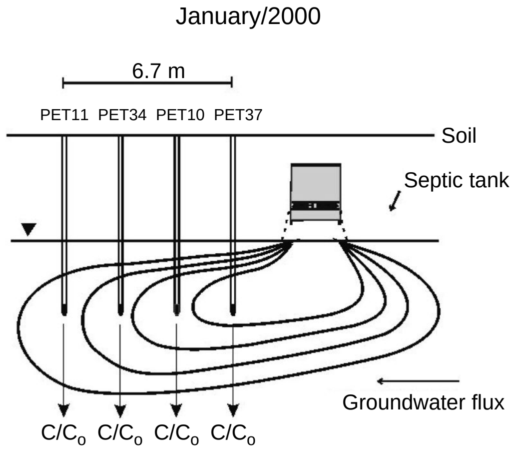
In the sodium nitrate-chloride water domain, the predominant cations were sodium and calcium, chloride and nitrate being the anions. The occurrence of these ions is clearly linked to contamination by effluents from the septic tanks.

The zoning of the contamination plume, from closer to more distant areas in relation to the septic systems, shows the following evolution of the nitrogenated series ions: (i) predominance of reduced forms (organic nitrogen + ammonium); (ii) ammonium + nitrate; and (iii) nitrate (Figure 5).

In the vicinity of the source of contamination (wells PET8, PET37, PET29, PET10 and PET34), the reducing environment is maintained by the degradation of organic matter from the septic tank itself. As one goes further away from the source of the contamination, more oxidized compounds, for example, ammonium and subsequently nitrate, appear due to reaction with waters that are richer in oxygen. Oxidation of organic matter and nitrification processes bring about the liberation of H^+ ions in solution, thus causing a drop in pH values. This can be described through Equations 2 and 3 below:

$$CH_2O + O_2 \Leftrightarrow CO_2 + H_2O \tag{2}$$

$$NH_4^+ + 2O_2 \Leftrightarrow NO_3^- + 2H^+ + H_2O \tag{3}$$



N_{org}	0.02	0.03	0.44	1
NH_4^+	0.40	4.00	6.20	1
NO_3^-	19.25	26.32	26.03	1
Na^+	0.17	0.73	0.70	1
Cl^-	0.18	0.24	0.69	1
pH	4.14	4.67	4.83	5.53 (absolute values)

Figure 5. Nitrogen compound zones and normalized concentration for various parameters (C/C_o) in the contaminant plume except for pH values (C_o = initial concentration at PET37).

Figure 5 shows percentage variation values (C/C_o) for the different nitrogenated compounds and major ions. It is based on a profile represented by wells PET37, PET10, PET34 and PET11 during January 2000. The concentrations measured at well PET37 were considered as C_o, due to the fact that this well is closer to the septic system (<0.5 m).

As expected, the chloride concentrations were highest in those wells located close to the septic tank, diminishing with distance from the source. As chloride is a conservative ion, this decrease is solely attributed to hydraulic dispersion processes.

In the same way as chloride, the concentration of other physical-chemical parameters decreases as one gets further from the source. The rate of decrease of these parameter concentrations cannot be explained solely by hydraulic dispersion. For example, concentrations of organic nitrogen underwent a decrease of 98 per cent and this indicates that, besides hydraulic dispersion, this species was subject to nitrification processes, giving rise to more oxidized compounds such as ammonium and, subsequently, nitrate. This is corroborated by the C/C_o values for these two ions (Figure 5).

Besides advection, the other cations and anions may be subject to the influence of adsorption. Similarly to chloride, sodium concentration showed a decrease of 83 per cent, indicating a greater influence of dispersion and a low degree of adsorption.

Electrical conductivity values resemble those of chloride. Increases in these values in groundwater reflect the transfer of ionic components from the septic system to the aquifer.

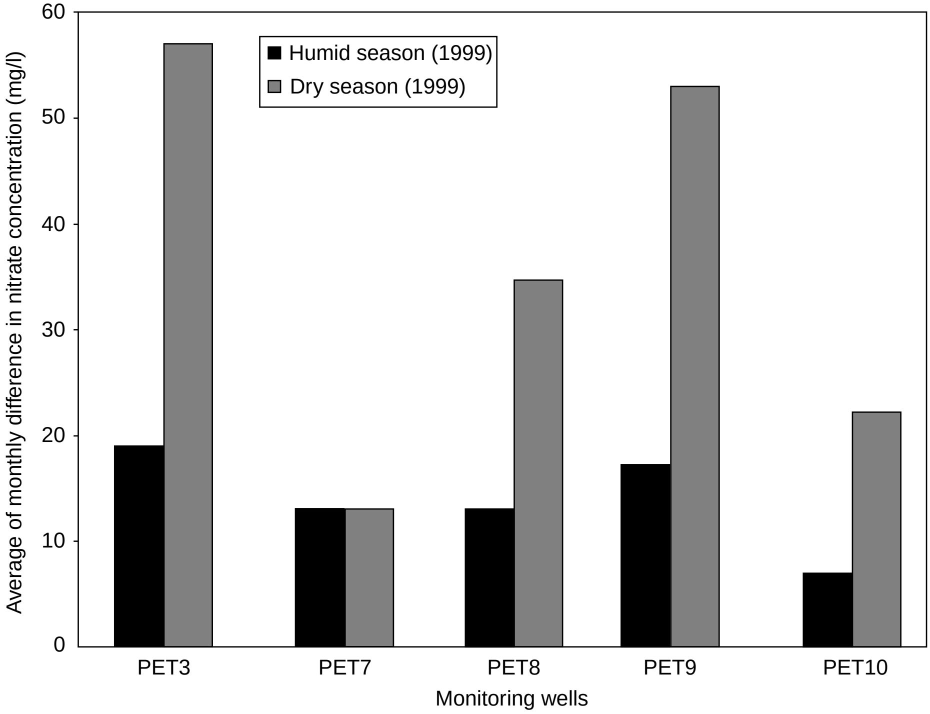


Figure 6. Mean values of monthly differences in nitrate concentrations at the monitoring wells (monthly average).

Temporal and spatial variations of monitoring well physical-chemical parameters

With regard to all the analyzed physical-chemical parameters, it was observed that, when considering all the sample wells, seasonal (rainy and drought seasons) variations in concentration at a particular well were greater than spatial variations during a specific season. For example, well PET3 showed nitrate concentrations ranging from 167 mg/l in September 1999 to 1.46 mg/l in January 2000, whereas the concentration ranges for all wells in the dry and wet seasons were $<1.33-112$ mg/l and $<1.33-167$ mg/l, respectively.

The nitrate concentration was measured in all monitoring wells monthly. Using the same approach as for the evaluation of water level seasonal variations (see Equation 1), the average of monthly differences in nitrate concentration was calculated for each monitoring well. Figure 6 presents the differences in concentration for wet and dry seasons.

A possible explanation for this variation in concentrations from the dry to the wet season in the same wells is that the recharge causes the deflection of the streamtubes reached by the monitoring wells.

During the dry season, the only recharge point is located at the contamination source. On the other hand, during the wet season the recharge occurs throughout the whole area, giving rise to a vertical flux component. This creates new clean water streamtubes and affects the ones that originate at the septic tank. Figure 7 illustrates these seasonal variations.

This hydraulic effect can be demonstrated through three-dimensional mathematical modelling using the Visual Modflow application (Guiguer & Franz, 1996). The modelled

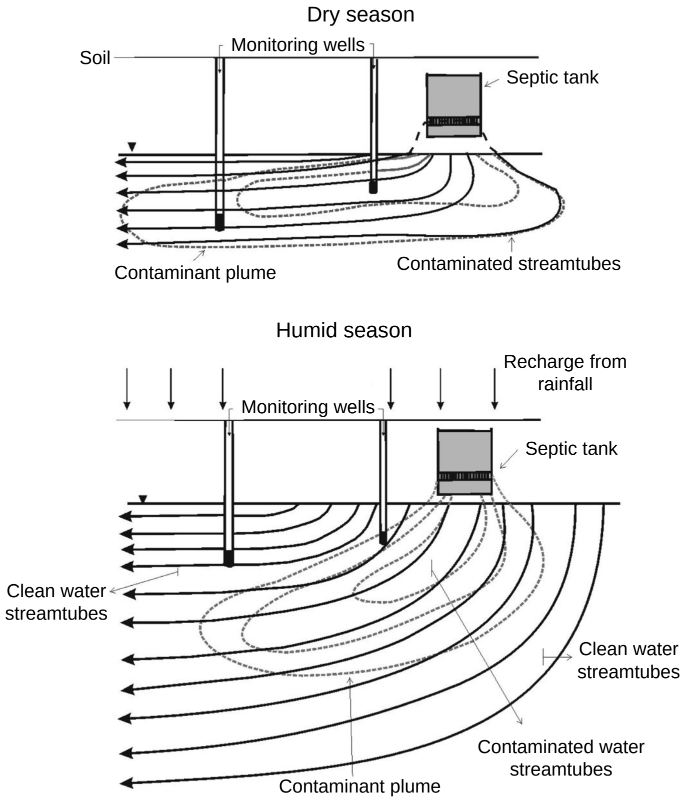


Figure 7. The effect of recharge in the behaviour of contaminant plume.

area was divided into 30 lines, 25 columns and four vertical layers. The number of layers adopted does not represent distinct hydrogeological systems, but increases the degree of precision obtained by the model. The hydraulic parameters for the four layers, namely specific yield (S_y), total porosity (n) and effective porosity (n_e), were defined based on the lithological characteristics of the formation and on hydraulic conductivity (K) data obtained through slug tests. The modelled area shows a region with a K value equivalent to 1×10^{-6} m/s, which is equal in all three directions ($K_x = K_y = K_z$) (Table 2).

The boundary conditions adopted were type I, assumed for the four layers (E-W limits), and type II, 'no flux', for the base of the model. Two recharge conditions were used including 90 per cent and 0 per cent total precipitation (scenarios 1 and 2) (Table 3). The model was run in transient condition. Ten particles situated to the west of the area were attributed to the model in order to visualize the behaviour of the flux lines and, consequently, of the streamtubes in the two scenarios considered (Figure 8). Transit time, shown between arrows, represents a period of 20 days. Modifications in transit time become more pronounced as the rate of recharge increases. This can be checked by comparing the tracks represented by dotted lines (scenario 1, no recharge) and those shown by continuous lines (scenario 2, 90 per cent precipitation recharge).

It is possible to demonstrate that the greater the recharge applied to the model, the greater the concavity of the flow lines and the greater the depth that they reach. During

Table 2. Hydraulic parameters used in three-dimensional mathematical modelling.

Hydraulic conductivity (m/s)	S_y	n_e	n
1.0×10^{-6}	0.20	0.15	0.25

Table 3. Recharge rates distribution in three-dimensional mathematical modelling scenarios.

Scenario	Period (days)		Recharge (mm/30 days)	Hydraulic head (m)	
	Begin	End		E area	W area
1	0	30	0.0	6.0	5.4
	30	60	0.0	5.5	5.0
	60	180	0.0	5.0	4.8
	180	210	0.0	4.8	4.2
2	0	30	207.0	6.0	5.4
	30	60	263.0	5.5	5.0
	60	180	82.0	5.0	4.8
	180	210	38.0	4.8	4.2

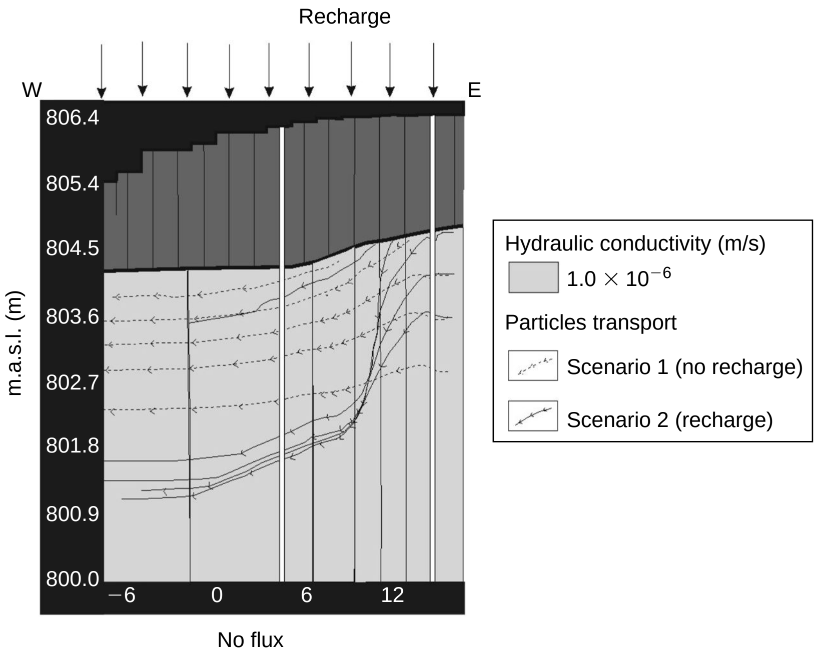


Figure 8. Particle tracking representation in recharge and no recharge scenarios.

the dry season, the flow lines are almost horizontal, thus reflecting absence of recharge. On the other hand, during the rainy season, streamtubes coming from the contaminant source, upgradient of the well zone, would be pushed downwards by the new streamtubes formed due to recharge (Figure 8).

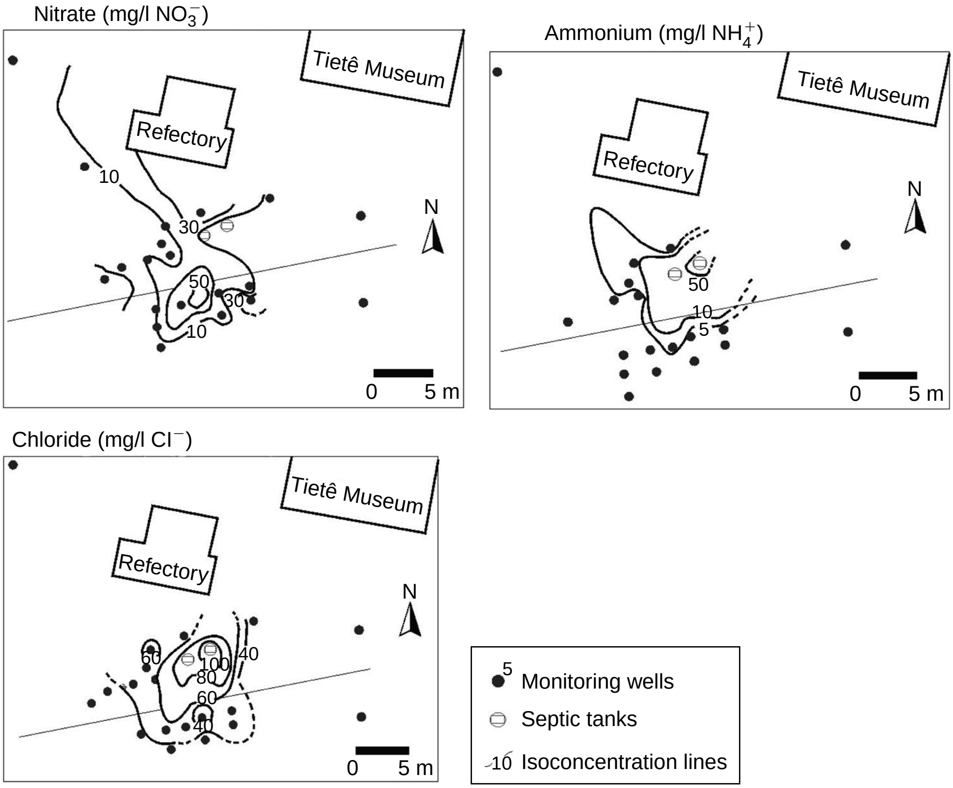


Figure 9. Nitrate, ammonium and chloride isoconcentration maps during dry season (August 1999).

Higher concentration 'islands' were present for such parameters as chloride and ammonium at several points (PET9, PET17, PET3, PET32 and PET27) for example (Figure 9). The occurrence of such 'islands' of nitrate concentrations was confirmed throughout the various sampling campaigns, by comparing them with the values obtained at nearby wells, considering a radius of influence of 3.5 m (Table 4).

When compared with the average concentrations found in neighbouring wells, the nitrate concentrations detected in wells PET9 and PET32 were lower in August 1999, similar or the same in September 1999, and higher in July 2000.

These points are characterized by lower hydraulic conductivity (K) values than the average value of their neighbours (K^*). Due to the lower K value, it would be more difficult for these points to be 'flushed' by the flux of water with different concentrations. Furthermore, the lower K values explain the deeper water levels observed in those 'islands'.

CONCLUSIONS

The nitrate concentrations detected in groundwater near the Tietê Museum, in the Tietê Ecological Park, Engenheiro Goulart Leisure Centre (PET-EG) were found to be above

Table 4. Hydraulic conductivity, water level and nitrate concentrations for an influence ray equal to 3.5 m, considering the period from July to September 1999.

PET	K (m/s)	K^* (m/s)	WL (m)	WL* (m)	NO_3^- (mg/l)	NO_3^-*(mg/l)
3	6.7×10^{-6}	8.2×10^{-6}	2.08	2.10	45	14
7	4.7×10^{-7}	4.7×10^{-6}	2.13	2.20	56	8
9	1.9×10^{-6}	8.3×10^{-6}	1.95	1.97	54	23
17	2.2×10^{-7}	3.5×10^{-6}	1.96	1.98	50	19
26	1.8×10^{-6}	8.4×10^{-6}	2.11	2.13	69	13
27	5.5×10^{-6}	1.1×10^{-6}	2.26	2.18	78	15
32	8.1×10^{-7}	1.7×10^{-5}	2.38	2.37	104	56

K: Hydraulic conductivity
K^*: Mean values of hydraulic conductivity at neighbouring wells
WL: Water level
WL*: Mean values of water level at neighbouring wells
NO_3^-: Nitrate concentration
NO_3^-*: Mean values of nitrate concentration at neighbouring wells

the legally established standard of 10 mg/l NO_3^--N. Evolution of a contamination plume, originating at the museum septic system, is controlled by the rapid hydraulics of the aquifer and also by nitrification phenomena.

Great variation in nitrogenated series concentrations and other physical-chemical parameters over time, especially during the dry season of the year, was observed at the site. This variation is caused by recharge and heterogeneity of hydraulic conductivity (K), which influence the streamtube network. Rapid recharge pushes the contamination plume downwards due to the income of non-contaminated water. Likewise, volumes of material of lesser hydraulic conductivity, when compared to the average K value in the vicinity (K^*), make it difficult to flush these points of the aquifer.

At this point, it should be emphasized that it was only possible to determine such phenomena due to frequent on-site monitoring and fortnightly measurements of some parameters, which made it possible to identify rapid variations in the chemical concentrations of pollutants in a single well. The time interval traditionally used in monitoring shallow unconfined aquifers would not be capable of providing this kind of detection and could lead to errors in interpretation. Based on this, it was concluded that a monitoring programme involving shallow aquifers with a rapid recharge rate and frequent sampling (weekly) should be conducted; otherwise there would be a risk of not detecting real concentrations in the contamination plume.

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CHAPTER 9

Natural and anthropogenic origin of chromium, nickel and manganese in groundwater in the Moa region (eastern Cuba)

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ABSTRACT: The aim of this study is to determine the possible sources of metals present in groundwater in the Moa region, Cuba. Rock, soil, water and metallurgical waste samples were collected in order to evaluate groundwater composition and origin. The hydrogeology of the area includes two aquifers (ophiolitic rocks and alluvial deposits). All the samples were analysed by inductively coupled argon plasma emission spectrometry and inductively coupled argon plasma mass spectrometry. According to its major components, groundwater in the ultramafic rock is of bicarbonated-magnesic type. In the alluvial aquifer two types of water are identified: magnesic-bicarbonated and sulphate-magnesic. The high values obtained for Cr(VI), Ni(II), Mn(II) and $Fe_{(total)}$ in groundwater can be explained by the existence of ultramafic rocks and nickeliferous lateritic deposits in which these metals are abundant. The high values of Cr(VI), Ni(II), Mn(II), $Fe_{(total)}$, SO_4^{2-} and Mg^{2+} in alluvial aquifers are due to polluted recharge from metallurgical waste of the tailing dam. The hydrochemical behaviour and migration of selected heavy metals (Cr(VI), Ni(II), Mn(II) and $Fe_{(total)}$) present in ultramafic rocks, lateritic profiles and the metallurgical waste have were studied with batch tests. These tests revealed the potential capacity of ultramafic rocks, the lateritic materials and metallurgical waste to transfer Cr(VI), Ni(II), Mn(II) and $Fe_{(total)}$ to groundwater. These results are good indicators for identifying the two sources of metals: natural and anthropogenic (by mine and metallurgical activities).

INTRODUCTION

The study is located in the catchment area of the River Moa, in northeastern Cuba, within the Moa mining district, province of Holguín (Figure 1). The climate of the region is humid subtropical and presents the following mean annual values: temperature of 24.5°C, relative humidity of the air of 85 per cent, rainfall of 2000 mm and evaporation of 1600 mm (INRH, 1986).

The region is characterized by the presence of an important mining industry that exploits the mineral resources from the lateritic deposit developed on top of the

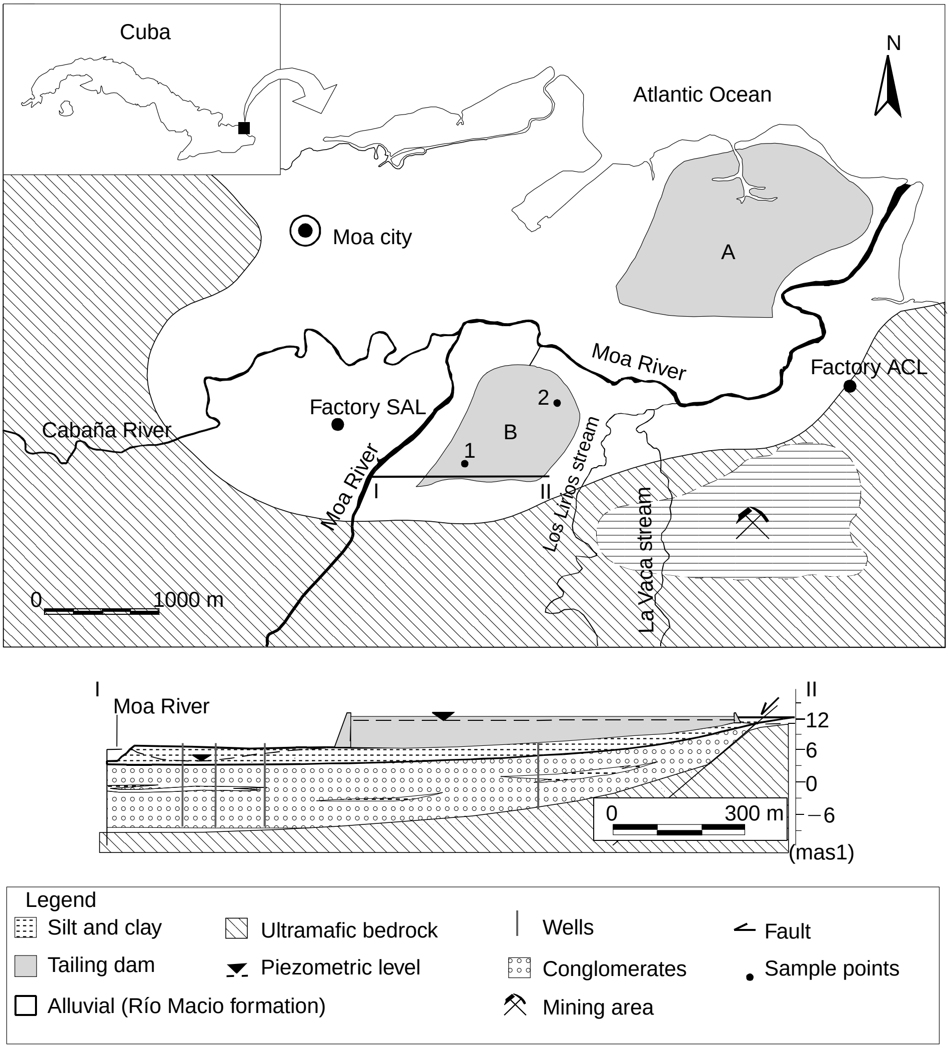

Figure 1. Geological scheme and profile I-II (INRH, 1986). SAL: lixiviation with sulphuric acid and ACL = Caron process. Tailing dams A and B store the metallurgical waste by ACL and SAL processes, respectively.

ophiolitic-related ultramafic rocks. Among these mineral resources, the Ni and Co reserves are considered one of the largest in the world. The exploitation of the lateritic mineral mass has been mined by the open-pit method in the Moa municipality since 1963. The lateritic deposit has mean contents of 1−2 per cent of nickel (Figure 1).

Two metallurgical processes for the extraction of nickel and cobalt are applied: (i) the Caron process (leaching with ammoniacal ammonium carbonate), from which nickel and cobalt oxides are obtained and (ii) lixiviation with sulphuric acid, from which nickel and cobalt sulphide are obtained. Both processes generate some 5200 tons of residual material per day. To date around 80 million tons have been accumulated in three tailing dams located on the terraces of the River Moa. The objective of this study is to determine the possible sources of metals detected in the groundwater in the alluvial and ultramafic rock aquifers, taking into account the geological materials and the metallurgical residuals present in the area.

MATERIALS AND METHODS

The water samples from the field campaign carried out in November 1996 were taken from springs located in ultramafic rocks (points 15, 22, 25 and 41−48) in the River Moa catchments and from wells in the alluvial aquifer (points 1−17) of the same river (Figure 2). Also, samples of the diverse lithologies present in the area were collected: (i) two samples of laterite profile in the Punta Gorda and Moa mines; (ii) one sample of each

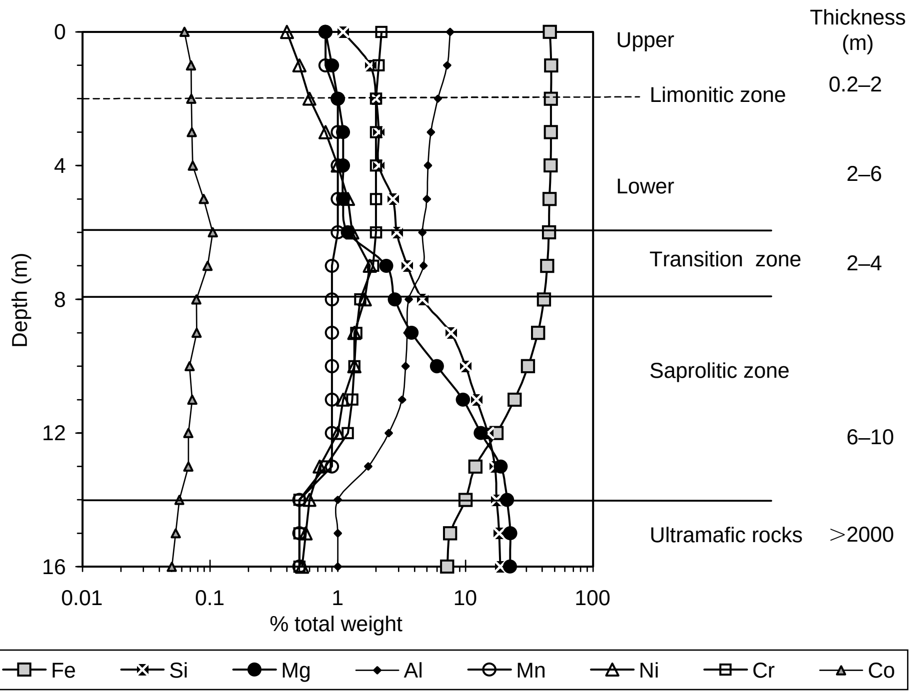

Figure 2. Lateritic profile and distribution of different elements vs. depth (Rodríguez-Pacheco et al., 2003).

lithologic type of the outcropped ophiolitic sequence (harzburguites, chromites, dunites, and gabbro); and (iii) two samples from the Rio Macio Formation. In addition, two samples from the solid and liquid residuals of the sulphuric acid process tailing dam were also collected.

To analyse the quality of waters, samples were collected from 29 points. Before the sampling the wells were purging. The water samples were filtrated (with 0.45 μm Millipore filters) and stored at 4°C. Two samples were taken at each point. A sample for analysis of the major elements and another was acidified with KNO_3 to preserve metals. An analysis of major elements in water samples was performed with methodologies widely used (Buurman et al., 1996). The total concentration of heavy metals was obtained through inductively coupled argon plasma emission spectrometry (ICP-AES) and inductively coupled argon plasma mass adsorption spectrometry (ICP-MS) for those cases in which concentration was much lower than 0.05 μg/l. Cr(VI) concentration was obtained through the colorimetric method.

The study of chemical transference to water was performed through lixiviation tests (batch) at a relationship of 1:10 in weight between solid material and water. The samples (dry metallurgic waste and geologic materials) were put in contact with Milli-Q water, with a pH of 5.5 and similar composition to rainwater in the study area, keeping them agitated for 24 hours at laboratory-controlled temperature (22 ± 2°C) (Table 1). In all water samples pH and Eh were controlled. The determination of Fe, Cr, Ni and Mn concentrations was performed with AAS and ICP-AES.

REGIONAL GEOLOGY

The study area is generally characterized by the presence of two types of material: (i) ophiolitic ultramafic rocks and (ii) sedimentary materials of the Rio Macio Formation (Figure 1).

The ultramafic rocks exposed in the study region are part of the Jurassic-Cretaceous Moa-Baracoa massif. These rocks outcrop on more than 60 per cent of the study area and have a thickness of up to 2.200 m. The ultramafic rocks correspond mainly to harzburgites (more than 70 per cent) and to a lesser extent to dunites. They have a variable degree of serpentinization that in shear-stress and fracture zones can reach up to 95 per cent of the whole rock (typical serpentinite). Besides these lithologies, 'impregnate peridotites' (with clinopyroxene and plagioclase), sills and dikes of gabbros have been reported, as well as podiform chromitites (Proenza et al., 1999a, 1999b).

Table 1. Physicochemical characteristics of rainwater in Moa and Milli-Q water used in the batch test.

	pH μS/cm	CE	NO_3	O_2(g)	CO_2	HCO_3^-	Cl^- (mg/l)	SO_4^{2-}	Ca^{2+}	Mg^{2+}	Na^+	K^+	Reference
Rain-water	4.86	10.66	0.04	5.33	0.27	1.42	1.77	0.56	0.45	0.16	1.17	0.08	Rodríguez-Pacheco, 2002
Milli-Q water	5.50	10.7	0.05	5.30	0.30	1.40	1.80	0.60	0.50	0.20	1.20	0.10	

The harzburgites mainly show porfiroclastic textures and consist of olivine (70–90 per cent by weight), orthopyroxene (8–20 per cent by weight), accessory chromites (1–2 per cent by weight) and clinopyroxene (up to 1 per cent modal). The dunites mainly show granoblastic textures and predominantly consist of olivine (between 96–98 per cent modal), and to a lesser extent of accessory chromite (up to 3 per cent) and orthopyroxene (<2 per cent by weight). These primary minerals are transformed to secondary minerals, basically serpentinite, clinochloride, magnetite, calcite, dolomite. To a small degree (<1 per cent modal) and associated to alteration, diverse Fe-Ni-Cu sulphides (pirrotine, pentlandite, heazlewoodite, millerite, calcosine) and Fe-Ni alloys (mainly awaruite) can be observed.

On top of the serpentinised ultramafic rocks a lateritic Fe-Ni-Co crust has been developed (it holds one of the biggest Ni-Co reserves of lateritic type in the world). In a typical cross-section of these lateritic deposits two main zones can be observed: a lower zone (saprolitic level) and an upper zone (limonitic level) (Figure 2). The saprolitic level consists mainly of minerals from the serpentine group, and to a lesser degree, of chlorite, ortopyroxene, talc, smectites, quartz, goethite and spinels (Ostromov et al., 1985; Rojas & Orozco, 1994). The limonitic level consists mainly of goethite, magnetite, maghemite, chromite and hematite. Also, to a lesser degree Mn oxides and hydroxides (asbolanes), gibbsite, chlorites and quartz can be found (Ostromov et al., 1985). The principle difference between the limonitec and saprolitic zones are the Ni, Co and Mg concentrations and the pH (Rodríguez-Pacheco et al., 2003). Figure 2 shows the distribution of different elements in lateritic profile according to weathering zone.

The Rio Macio Formation (Holocene), with an extension of 10 km^2, consists of alluvial deposits on the terraces of the River Moa (Figure 1). The minerals in the alluvial material are goethite, hematite and, to a lesser extent, gibbsite and serpentine (INRH, 1986).

HYDROGEOLOGICAL MODEL

In the study area two unconfined aquifers can be distinguished: the serpentenised ultramafic and mafic rocks (harzburgites, dunites and gabbro); and the alluvial deposits. On the top of the ultramafic rocks there is a lateritic crust with thickness varying between 1 and 20 meters. The upper part of the lateritic profile, where it exists, corresponds to the unsaturated zone of this aquifer. According to previous hydrogeological studies the piezometric levels in the ultramafic rocks can vary seasonally (up to 10 m) (Rodríguez et al., 1999; Rodríguez-Pacheco, 2002), indicating a low storage coefficient. On the other hand, in the alluvial aquifer the levels fluctuation is much lower (Table 2). The

Table 2. Hydrogeologic parameters (INRH, 1986; Terrero, 1986).

Parameters	Alluvial	Peridotites	Laterites
Permeability (m/d)	28–134	1–14	0.15–5.6
Transmissivity (m^2/d)	700–3350	30–840	0.13–4.2
Hydraulic gradient (i)	0.02–0.026	0.03–0.08	0.023–0.06
Total porosity (per cent)	15–30	3–8	20–60
Saturated thickness (m)	25–30	30–600	1.5–20

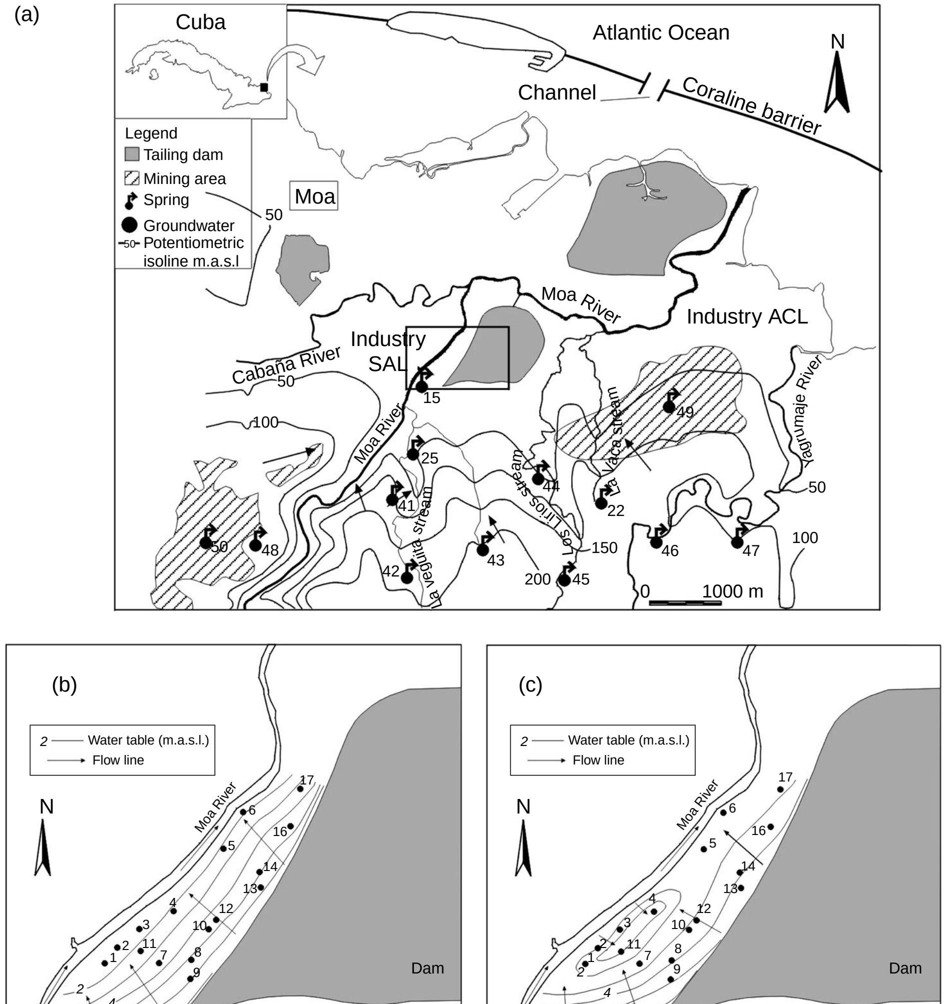

Figure 3. (a) Isopiestic map of the study area (adapted from INRH, 1986); (b) Isopiestic maps of the alluvial aquifer in steady state flow (November, 1986) and (c) Isopiestic maps of the alluvial aquifer in transitory state flow (November, 1996).

predominant flow direction is SW-NE and the isopiestic lines reproduce the topography of the terrain (Figure 3).

The aquifer recharge for both aquifers is due to the infiltration of rainwater and it is estimated in 400 mm/year (Rodríguez et al., 1999). In the alluvial aquifer also there is a recharge induced by the tailings dam and the pumping from the River Moa, where $8 \times 10^6\,m^3$/year are extracted for the supply of some 75,000 habitants and industry (Figure 3). The pumping in the alluvial aquifer changes the slope in the piezometric level below the level of the river, which causes an induced recharge.

CHEMICAL CHARACTERISTICS OF GROUNDWATER

Turbidity: the ultramafic rock aquifer shows values lower than 1 ppm of SiO_2, whereas the alluvial aquifer has from 1–2 ppm of SiO_2 (Table 3). These values indicate the presence of colloidal material (e.g. Custodio, 1983; Rodríguez-Pacheco, 2002).

Electrical conductivity: in the ultramafic rock aquifer it varies from 100–500 μS/cm and for the alluvial aquifer from 200–7.300 μS/cm. The higher values of conductivity are relate to the samples from wells 12, 13, 14, 16 and 17 located near the tailing dam (Table 3).

pH: values for the ultramafic rocks and the alluvial aquifer range from 6.7–8.2. Within the alluvial aquifer the low values were obtained from wells near the tailing dam (Table 3).

Alkalinity: in the ultramafic rock aquifer this varies from 1–5 meq/l expressed in CO_3Ca. In the alluvial aquifer it varies from 1–6 meq/l, augmenting its values, as one gets closer to the tailing dam (Table 3).

Hardness: this is mainly due to the presence of magnesium. In the ultramafic rock aquifer Ca + Mg content is between 1 and 3 meq/l; in the alluvial aquifer it varies widely from 1.7–103 meq/l, being higher in wells near the tailing dam (Table 3).

Dissolved oxygen (DO): in both aquifers the water content of DO is higher than 1.5 mg/l. According to the water oxygen content they can be considered to be oxidising media (Table 3).

Chemical oxygen demand (COD): in the alluvial aquifer this can be considered to be low, varying from 0.2–0.7 mg/l (Table 3), which indicates that there is hardly any metal or organic matter content that could be oxidized (e.g. Custodio, 1983).

Total dissolved solids (TDS): in the ultramafic rock aquifer the concentrations are low (between 50 and 250 mg/l). In the alluvial aquifer they increase as one gets closer to the tailing dam; TDS is higher than 2000 mg/l at points 12–17 (Table 3). Sometimes, the TDS shows values of more than the permissible drinking water standards (ADWS), TDS > 2000 mg/l, as recommended by the World Health Organization (1995 in Rodríguez-Pacheco, 2002).

Calcium (Ca^{2+}): concentration in the ultramafic rock aquifer is very low (between 3 and 4 mg/l). In the alluvial aquifer it is highly variable, between 14 and 201 mg/l (Table 3). In the water contained in ultramafic rocks its presence is probably due to metheorization impregnated peridotites (plagioclase-bearing peridotite) and gabbro bodies (e.g. Rodríguez-Pacheco, 2002; Proenza et al., 1997).

Magnesium (Mg^{2+}): concentration of Mg in the ultramafic rock aquifer is low, 9–10 mg/l, whereas in the alluvial aquifer it varies from 26–1114 mg/l. At points 8, 9, 10, 12, 13, 14, 16 and 17 the mean Mg content widely exceeds the limits for drinking water (Mg > 250 mg/l), recommended by the WHO (Table 3). In the ultramafic rock aquifer its presence is due to the alteration of olivine (forsterite content > 90 per cent) and orthopyroxene (enstatite content > 90 per cent) (e.g. Proenza et al., 1997).

Sodium (Na^{+}): concentration in the ultramafic rock aquifer is 6–7 mg/l. In the alluvial aquifer it varies between 4 and 37 mg/l (Table 3). Its origin is atmospheric precipitations. This element is concentrated by the evaporation that takes place in the area (e.g. Custodio, 1983; Rodríguez-Pacheco, 2002; Proenza et al., 1997).

Potassium (K^{+}): concentration in the ultramafic rock aquifer is less than 1 mg/l. In the alluvial aquifer its concentration is generally less than 1.5 mg/l. Its origin is in rainfall

Table 3. Characteristic groundwater in alluvial and ultramafic aquifers (T = turbidity, CE = electrical conductivity).

Point	pH	T (ppm SiO_2)	CE (µS/cm)	NO_3	O_2(g)	DQO	Cr^{+6}	Mn^{2+}	Ni^{2+}	$Fe_{(total)}$	SiO_2 (mg/l)	HCO_3^-	Cl^-	SO_4^{2-}	Ca^{2+}	Mg^{2+}	Na^+	K^+
1	7.4	0.23	218	1.71	2.10	0.04	0.02	2.32	0.01	0.48	13.0	126.5	15.0	6.0	8.7	26.5	4.2	1.0
2	7.4	0.56	280	3.00	1.53	0.00	0.02	2.64	0.03	1.34	13.1	145.0	13.5	16.5	9.0	30.0	4.7	1.5
3	7.4	0.52	210	3.29	1.52	0.00	0.02	2.49	0.01	0.60	12.8	118.5	16.0	16.5	13.6	22.8	4.1	1.0
4	7.2	0.36	600	2.15	1.48	0.68	0.02	2.26	0.02	0.52	12.9	274.5	15.0	77.5	11.9	64.5	6.1	1.0
5	8.1	0.56	600	2.00	1.60	0.00	0.02	2.67	0.01	0.46	12.9	310.5	15.0	49.5	14.7	63.5	8.7	1.0
6	7.3	0.58	732	1.55	2.40	2.99	0.02	2.61	0.01	1.40	15.6	314.5	19.5	100.0	13.5	79.5	9.6	1.0
7	7.6	0.92	650	3.81	1.65	0.00	0.05	4.04	0.03	2.30	17.8	156.5	23.5	178.0	14.0	67.0	13.5	1.0
8	7.1	0.74	1520	1.40	1.50	0.69	0.06	5.77	0.04	3.16	16.6	228.0	27.0	433.0	76.0	98.3	18.6	1.0
9	7.1	0.95	1750	10.08	3.18	2.11	0.40	6.10	0.05	3.91	16.1	189.0	24.0	639.0	23.0	182.8	11.7	1.0
10	7.5	0.86	1708	1.00	1.95	0.00	0.06	5.53	0.04	2.38	16.0	189.0	20.5	632.2	18.5	182.5	11.5	2.0
11	7.3	0.46	550	1.94	1.15	0.48	0.04	3.19	0.03	2.03	27.6	365.0	33.5	40.0	14.0	58.0	17.5	1.0
12	7.2	1.23	5310	11.52	6.30	0.91	1.11	7.05	0.06	4.10	15.5	384.0	51.5	1824.0	49.0	433.8	214.4	1.0
13	7.3	1.24	4402	1.26	2.12	0.03	1.09	7.71	0.06	4.29	16.4	381.0	45.5	3512.0	176.0	756.6	75.5	1.5
14	6.8	2.01	4612	3.96	1.75	1.04	1.41	7.80	0.06	3.73	26.1	279.0	44.0	3072.0	115.7	748.8	35.5	1.5
16	6.7	2.14	5915	3.10	2.28	1.66	1.60	8.08	0.08	5.04	27.0	402.5	51.5	4771.5	173.5	1144.5	37.5	1.0
17	6.9	2.10	3440	12.40	3.70	1.29	0.78	6.88	0.06	4.35	26.1	421.0	32.5	2489.5	201.8	581.0	23.0	1.5
15	7.1	0.25	54.7	9.70	1.52	0.51	0.008	0.005	0.005	0.07	10.5	35.9	16.2	1.42	2.8	9.2	6.2	1.1
22	6.5	0.60	50.8	11.20	1.61	0.32	0.006	0.004	0.005	0.07	8.3	35.5	16.2	1.54	2.8	9.2	6.2	1.1
25	6.2	0.65	51.3	10.10	1.91	0.62	0.006	0.005	0.007	0.08	7.6	36.6	14.2	1.55	3.1	8.9	7.0	1.0
41	6.3	0.78	56.8	9.20	1.60	0.58	0.002	0.003	0.004	0.08	8.1	39.5	16.2	1.64	2.8	9.1	6.9	0.21
42	6.6	0.81	41.3	9.80	1.71	0.47	0.002	0.004	0.003	0.09	7.9	46.6	14.2	1.35	3.1	9.9	7.9	0.5
43	6.7	0.45	45.7	12.20	1.53	0.33	0.003	0.003	0.005	0.06	8.5	65.5	16.2	1.74	2.8	9.2	6.4	0.3
44	6.8	0.46	55.2	13.30	1.57	0.29	0.001	0.002	0.007	0.07	7.9	56.6	14.2	1.85	3.1	8.1	5.0	0.2
45	7.1	0.65	58.1	11.80	2.01	0.31	0.002	0.003	0.005	0.05	8.8	45.1	16.2	1.34	2.8	7.7	6.8	0.6
46	7.1	0.32	61.3	9.00	1.55	0.18	0.003	0.005	0.007	0.04	9.6	32.9	14.2	1.51	3.1	8.5	7.3	0.5
47	6.9	0.68	50.8	10.60	1.58	0.50	0.005	0.003	0.005	0.06	8.3	37.1	16.2	1.44	2.8	7.6	6.5	0.7
48	7.1	0.83	51.3	11.30	1.53	0.22	0.002	0.005	0.007	0.08	6.5	39.6	14.2	1.35	3.1	9.1	7.0	0.8
49	7.2	1.25	66.2	8.60	2.01	0.26	0.008	0.005	0.005	0.17	10.5	51.1	16.2	1.64	2.8	19.9	12.2	0.2
50	7.3	1.36	71.9	7.70	2.04	0.31	0.007	0.005	0.007	0.82	12.3	54.9	14.2	1.45	3.1	18.7	14.0	0.1

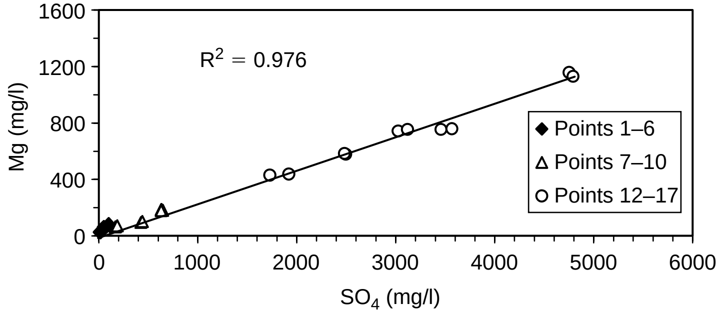

Figure 4. Relationship between the concentration of sulphate and magnesium in the alluvial aquifer.

(0.06–0.08 mg/l) (Table 3). These low concentrations are due to ionic exchange with the clay materials of the lateritic profile and the sediments of alluvial aquifer during rainfall infiltration (e.g. Custodio, 1983; Rodríguez-Pacheco, 2002).

Sulphate (SO_4^{2-}): sulphate concentration in the ultramafic rock aquifer is from 3–9 mg/l. The concentration of sulphates in the alluvial aquifer varies widely from 200–4800 mg/l. At points 8, 9, 10, 12, 13, 14, 16 and 17 the mean sulphate content widely exceeds the values of ADWS, set at 400 mg/l (Table 3). In the ultramafic rock aquifer its origin is rainfall and concentration through evaporation (e.g. Custodio, 1983; Rodríguez-Pacheco, 2002). Small amounts of sulphate might come from oxidation of sulphides present in peridotites, whereas in the alluvial aquifer concentrations are the result of infiltration from the tailing dam. The high content of sulphate is favoured by the presence of high concentrations of Mg, which substantially increase solubility (e.g. Custodio, 1983). The relationship Mg vs. SO_4^{2-} is nearly lineal (Figure 4).

Bicarbonate (HCO_3^-): in the ultramafic rock aquifer, its concentration is generally less than 200 mg/l, although occasionally concentrations of up to 500 mg/l are present. In the alluvial aquifer the concentration is variable (118–421 mg/l), with increasing concentrations as one gets closer to the tailing dam (Table 3). The presence of HCO_3^- is due to hydrolysis of the silicates of the ultramafic rocks and atmospheric CO_2 (e.g. Custodio, 1983; Rodríguez-Pacheco, 2002).

Chloride (Cl^-): concentration in groundwater in the ultramafic rock varies from 11–18 mg/l, whereas in the alluvial aquifer it varies from 13–51 mg/l (Table 3). The origin of chloride ions in both aquifers is rainfall and concentration is associated to evaporation.

The presence of higher concentrations of all dissolved ions near the tailing dam is due to the induced recharge of lixiviation products. According to its major components, water in the ultramafic rock is of bicarbonated-magnesic type. In the alluvial aquifer two types of water are identified: magnesic-bicarbonated (points 1–6) and sulphate-magnesic group (points 7–17) affected by the recharge from the tailing dam (Figure 5).

METALS IN GROUNDWATER

In this section we analyse those elements of natural origin that are present in groundwater and sometimes have higher concentrations than permissible drinking water standards,

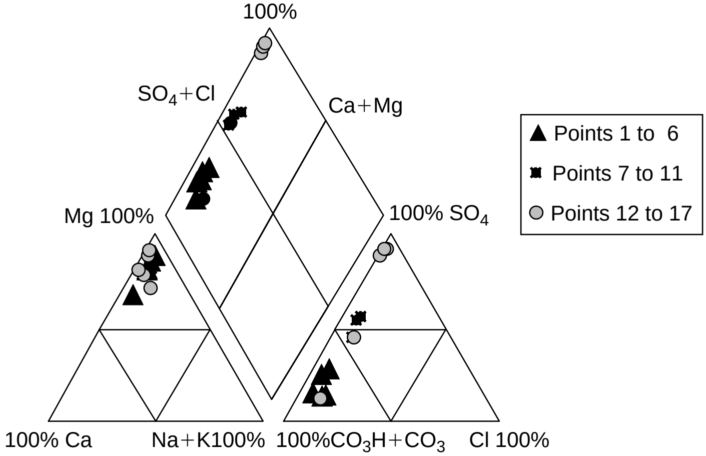

Figure 5. Piper diagram for water in the alluvial aquifer.

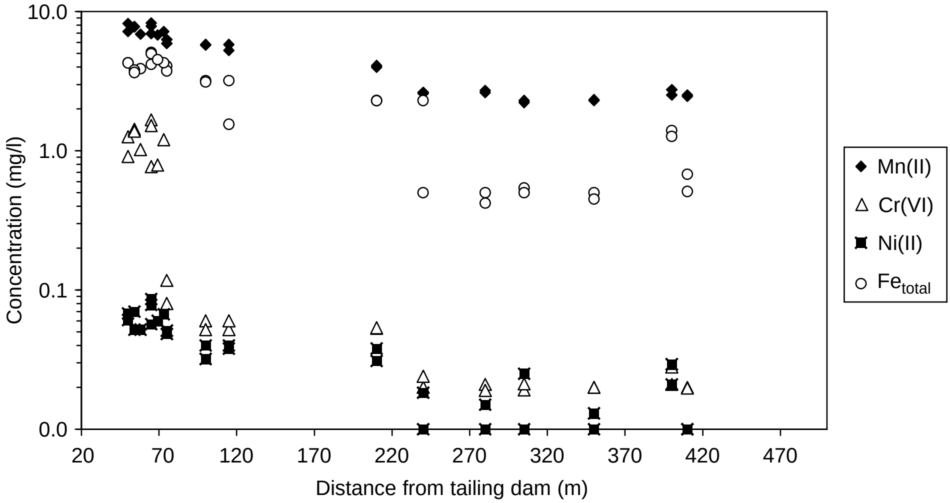

Figure 6. Variation in concentrations of the main contaminants in the alluvial aquifer vs. distance to the tailing dam.

recommended by the WHO (1995 in Rodríguez-Pacheco, 2002). With respect to metals, it is especially interesting to highlight the concentration of Cr, Mn, Ni and Fe in the alluvial aquifer (Figures 5 and 6), given their association with mineralisation of the residues and geology of the area.

Chromium (Cr): concentration of hexavalent Cr(VI) in the ultramafic rock aquifer varies from 0.001–0.002 mg/l. The presence of Cr(VI) in the alluvial aquifer has been detected in all samples with values from 0.01–1.60 mg/l. Concentration of Cr(VI) exceeds the ADWS limit (0.05 mg/l) at points 8, 10, 11, 12, 13, 14, 16 and 17. Its presence is the result of lixiviation products infiltrating from the tailing dam and its mobility is possibly enhanced by an oxidising medium in the aquifer. Chromium may be adsorbed at the surface of Fe and Mn oxides in colloidal form (amorphous compounds) that are present in the matrix of the porous media and in the water of the aquifer. Solubility of Cr(VI) is mainly controlled by pH values (e.g. Weng et al., 1994). As we go further away from the tailing dam pH in the alluvial aquifer increases up to 8.2 and Cr(VI) concentration decreases to 0.01 mg/l. These results are coherent with those obtained by other authors (e.g. Vardaki & Kelepertsis, 1999; Whalley et al., 1999; Robles-Camacho & Armienta, 2000), who have detected presence of Cr in aquifers located in areas where ultramafic rocks and their associated meteorised products exist.

Nickel (Ni): in the water of ultramafic rock aquifer this is present in low concentrations with a mean value of 0.01 mg/l. In the alluvial aquifer concentrations increase in the opposite direction of the flow lines with values from 0.01–0.09 mg/l. Ni concentration at points 12, 13, 14 and 17 exceeds the ADWS limit (0.05 mg/l). The natural origin of Ni in groundwater within ultramafic rocks has been described by several investigators (Kudelasek & Zamarsky, 1971; Formell & Oro, 1980; Candela & Rodríguez, 1996; Vardaki & Kelepertsis, 1999) and in all cases they report concentrations lower than 0.02 mg/l. Solubility of Ni is limited to pH values over 6. An explanation of this could be that part of Ni moves in association with the colloidal particles (e.g. Fernández-Aller, 1981).

Manganesium (Mn): concentration in the ultramafic rock aquifer is from 0.04–0.05 mg/l. In the alluvial aquifer concentrations, it varies from 2.1–8.3 mg/l. All samples from the alluvial aquifer exceed the ADWS (0.5 mg/l). Since solubility of Mn is limited to pH > 6 (e.g. Fernández-Aller, 1981; Weng, et al., 1994), Mn concentration can probably be related to colloidal particles (amorphous material).

Iron (Fe_{total}): concentration of total iron in the ultramafic rock aquifer is below 0.08 mg/l. In the alluvial aquifer its varies widely, for Fe^{2+} from 0.035–0.44 mg/l. Total iron is present in concentrations from 0.035–5.1 mg/l. Concentration of total iron at points 7, 8, 9, 10, 11, 12, 13, 14, 16 and 17 exceeds the ADWS (1 mg/l).

Analysis of relationships among diverse contaminants shows that: sulphate and total metals concentration shows an exponential correlation with distance of the sampling point to the tailing dam (Figure 7b). Ni vs. Fe and Mn vs. Ni show similar behaviour (Figure 7a and 7c). According to the results shown in Figure 7, it is possible to distinguish three groups of samples: (i) wells closer to the tailing dam (12, 13, 14, 16 and 17); (ii) wells located in the area of influence of pumping wells and induced recharge from the dam (7, 8, 9 and 10); and (iii) wells located near the River Moa (1–6) (Figure 7d). The most evident difference between these groups is in Cr-Mn concentration vs. distance to the tailing dam relationship (Figures 6 and 7d).

Sampling points with the highest sulphate concentrations have also the highest concentrations of Ni, Cr, Mn and Fe. It can be also observed that a relationship exists between sulphate and the sum of many metals present in groundwater (Figure 7c). For all the sampled points two groups may be differentiated: points near the tailing dam (8, 9, 10, 11, 12, 13, 14, 16 and 17) and points near the pumping zone (1 to 7). The fact that the metals dissolved by acidic waters, which infiltrate from the tailing dam, might be due either to groundwater having not reached enough alkalinity for precipitation or to their association with colloidal particles present in the aquifer.

ORIGIN OF METALS PRESENT IN GROUNDWATER

Figure 8 shows the results from batch tests carried out on rocks, laterite and metallurgical waste (relation solid/liquid 1:10). In all cases it is possible to appreciate the capability of natural and metallurgical waste to lixiviate metals when it is put in contact with water. The metallurgical wastes present the highest values of mass of metals lixiviated. The existence of two sources of metals, natural and anthropogenic origin, can be deduced from these results.

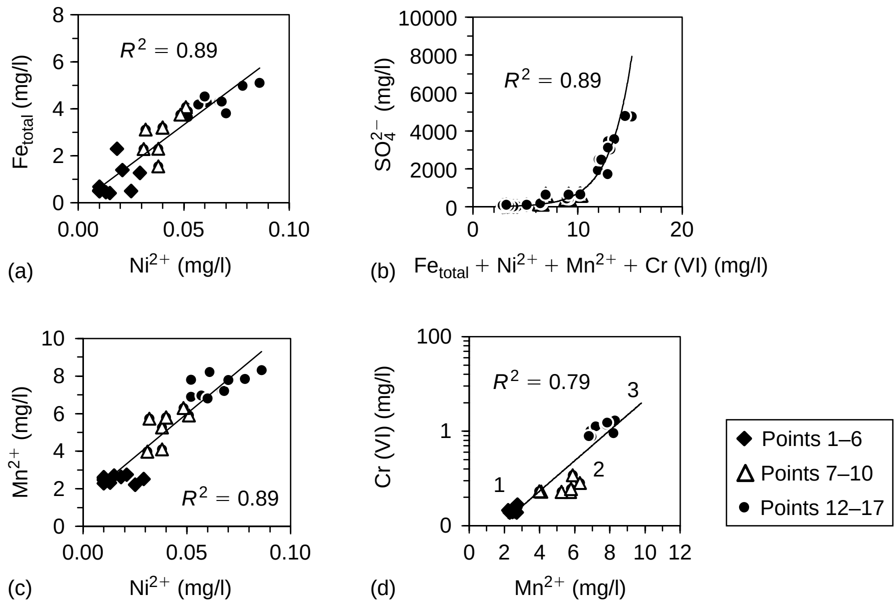

Figure 7. Relationship between diverse contaminants in the groundwater of the alluvial aquifer: (a) Ni vs. manganese, (b) sulphate vs. total metals, (c) nickel vs. iron and (d) manganese vs. chromium.

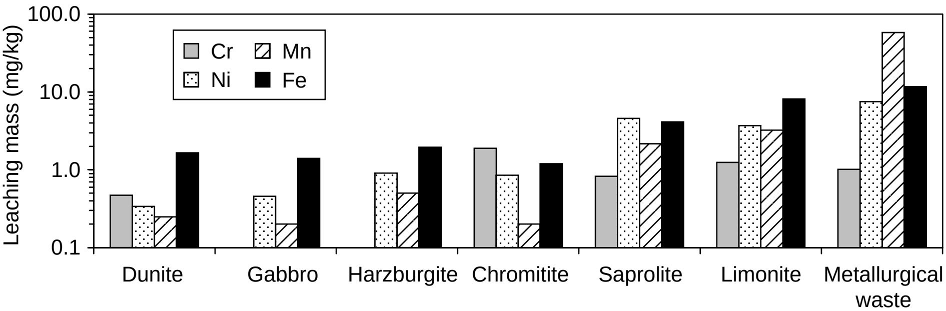

Figure 8. Results of the batch tests for metallurgical waste, rocks and laterites profile of the study area.

Natural sources of Cr, Ni, Mn and Fe

Among the four metals with concentrations that sometimes exceed ADWS (0.05 mg/l), Cr(VI) is of special interest because it represents the major environmental risk with its toxicological effects on human beings. Ni presents some toxicity for concentrations of more than 0.05 mg/l.

Chromium: the ultramafic rocks, compared to other types of rocks, contain relatively high amounts of Cr. In the study area, the ultramafic rocks and their alteration products (lateritic profile) are the most abundant lithologies, possibly constituting a natural source of Cr.

The main minerals that compose the ophiolitic rocks are olivine, orthopyroxenes, chromite, clinopyroxenes and plagioclase. Among all these phases, those that contain the highest amount of Cr are, in decreasing order, chromite, clinopyroxenes and orthopyroxenes. Chromite in the chromitite, and accessory chromite within harzburgite, dunite and gabbro shows values of Cr_2O_3 from 36–46 per cent by weight. Clinopyroxene shows values of Cr_2O_3 from 0.90–1.53 per cent by weight, whereas orthopyroxene varies from 0.47–0.54 per cent by weight. However, olivine and plagioclase are phases poor in Cr and their contents are normally below the detection limit of electronic microprobe (Proenza et al., 1997). Thus, we infer that the natural source of Cr is associated with those rocks that show the highest proportion of Cr-rich mineral phases (chromite and pyroxene).

The results obtained from batch tests indicate that the rock type that transfers most Cr to the water is chromitite, followed in decreasing order by dunite, gabbro and harzburgite (Figure 8). The highest partition values of Cr from the chromitite sample agree with its mineralogical composition, more than 90 per cent chromite. However, it seems paradoxical that harzburgite and gabbro, which contain an important modal proportion (per cent of mineral) of chromiferous pyroxene, transfer less Cr to water than the dunite, which does not have pyroxene in its composition. The answer to this question is probably related to the different physicochemical behaviours of chromite and pyroxene. Systematically, chromites present in the studied samples have a ferrichromite alteration around grains and fractures (Proenza et al., 1997). This alteration of chromite is characterized by a considerable increase in Fe^{3+} and it is very developed in chromitites and dunites, but much less developed in harzburgites. In the latter, accessory chromites are usually within orthopyroxenes, remaining much less altered (Proenza et al., 1997, 1999a). The results obtained in this study show a direct correlation between the content of Cr in water and the degree of alteration to ferrichromite of the chromite crystals that compose the diverse analysed lithologies. These results agree with the conclusions obtained by Robles-Cacho and Armienta (2000). These authors suggested that disintegration of the ferrichromite borders, due to their reduced physical-chemical stability, is the main geochemical process that incorporates Cr to aquifers in ultramafic rocks at Sierra de Guanajuato (Mexico).

In the analysis of the samples corresponding to the lateritic crust, both revealed the transfer of similar amounts of Cr, with slightly higher amounts given by the sample in the limonitic level. In the lateritic profile chromites have an important degree of alteration to ferrichromite (e.g. Friedrich et al., 1987), which constitutes an important source of Cr. Additionally, these lateritic samples are characterized by important modal proportions of goethite (up to 60 per cent in the limonitic zone), which can potentially transfer Cr to water. The goethite contain important amounts of Cr in its structure; for example Manceau et al. (2000) report values of up to 0.73 per cent by weight Cr in natural goethite.

Nickel: among minerals present in the lateritic profile on top of the ultramafic rocks it is goethite that contains the highest concentrations of Ni (up to 1.3 per cent by weight) (Golightly, 1981; Rojas & Orozco, 1994). In addition ultramafic rocks have olivine (0.4–0.5 per cent by weight) and serpentine (0.2–0.3 per cent by weight) and an important percentage of Ni. According to the batch test results, those samples that transfer the highest percentage of Ni to water correspond to the saprolytic zone where Ni content can reach 3 per cent by weight (e.g. Golightly, 1981; Friedrich et al., 1987; Rojas & Orozco, 1994) and 0.3–1.5 per cent by weight in limonitic zone. In the case of the analysed rocks it is harzburgite, with a NiO content from 0.2–0.3 per cent by weight, that releases the highest amounts of Ni (Figure 6). Part of the concentration of Ni detected in water from the ultramafic rock aquifer might be due to oxidation of disseminated Ni sulphides present in ultramafic rocks. These results are coherent with the studies of Kudelasek & Zamarsky (1971), Formell & Oro (1980) and Candela & Rodríguez (1996), where Ni contents were detected in groundwater of ultramafic rock aquifers with $pH > 7$ and $Eh > 0.1$.

Manganese: concentration of manganese in ultramafic rocks is in the order of 0.01 per cent by weight, whereas in the upper part of the alteration material that forms the lateritic crust it may reach 0.12 per cent by weight (Golightly, 1981). The existence of manganese in groundwater of the ultramafic rock aquifer might have its origin in the amorphous manganese content in the lateritic crust. The presence of amorphous manganese in the lateritic profile of the Moa region has been reported by several authors (Rojas & Orozco, 1994; Almaguer, 1995; Olivera et al., 2001).

Iron: in the geologic material studied from both aquifers, iron is one of the most abundant elements with contents ranging from 9 per cent by weight of Fe_2O_3 in the host rock to 77 per cent by weight in materials conforming to the lateritic profile (Rojas & Orozco, 1994; Almaguer, 1995). Its natural origin in ultramafic rock aquifers is generally related to the dissolution of ferric silicates (e.g. Custodio, 1983), although in these cases it is probable that part of this iron (Fe^{2+}) content is due to the oxidation of low concentrations of sulphide disseminated in the ultramafic rocks. Concentration of Fe detected in the aquifer of ultramafic rocks is coherent with results obtained by Kudelasek & Zamarsky (1971) and Formell & Oro (1980). In the water that flows through the ultramafic rocks of the Moa region the presence of Fe^{2+} (0.1 mg/l) and Fe^{3+} (0.15–1.5 mg/l) has been reported for pH from 7.1–7.4 (Formell & Oro, 1980).

Anthropogenic source of Cr, Ni, Mn and Fe

The anthropogenic source of the four metals studied is related to the infiltration of leachates from the tailing dam, which have an acid pH. The solid phases in the tailing dam are composed of different minerals such as hematite (69–75 per cent), Cr-spineles (2.1–2.8 per cent), gibbsite (1.4–6 per cent), gypsum (2.5–5.6 per cent) and magnetite (2–3 per cent). The concentration of major elements is: 47 per cent Fe; 0.48 per cent Mn; 0.08 per cent Ni; 0.011 per cent Co; 4.3 per cent Al; 0.044 per cent Mg; 0.042 per cent Cu; 0.05 per cent Zn; 1.65 per cent Cr. Pore water from the metallurgical residue that infiltrates through the base of the dam has a pH of 4.1 and the following average concentrations of dissolved salts: 4,250 mg/l SO_4^{2-}; 120 mg/l Mn^0; 2,220 mg/l Mg^{2+}; 530 mg/l Ca^{2+}; 36 mg/l Na^+; and 1.67 mg/l Cr(VI).

The fact that metals in the aquifer are in solution is due to a low pH, low alkalinity and an oxidizing environment. As the distance of the sampling points to the tailing dam increases, the contamination decreases exponentially until a distance of 250 m is reached. For greater distances concentration of metals is relatively constant, indicating the presence of a proportionally high background content in the alluvial aquifer (Figure 6 and Table 3).

CONCLUSIONS

Considering the geological context, results from laboratory analysis, regional climatic conditions and the characteristics of metallurgical waste, it is possible to differentiate two sources of metals: one of natural origin due to weathering processes in ophiolytic rocks, which produced the mobilization a different elements, and an anthropogenic source due to the infiltration of leachates from the tailing dam.

Based on the concentrations of Cr, Ni, Mn and Fe in the water of the ultramafic rock aquifer, we conclude that the background of the region is high, because this area is not affected by mining activity. Concentration of Mn is above the limit for drinking water standard recommended by WHO (1995 in Rodríguez-Pacheco, 2002). The rest of the elements have to be taken into account because small increases in Cr and Ni due to mining activity might make the groundwater unsuitable for drinking.

As a result of the batch tests, we conclude that most of the samples from geologic materials and metallurgical waste have the ability to transfer diverse proportions of soluble salts to groundwater, among which important proportions of metals (Cr, Ni, Mn and Fe) are present.

Mobilization of different metals in the alluvial aquifer is a function of: (i) characteristics of the contaminant source, (ii) hydrogeochemical characteristics of the porous media; and (iii) residence time of contaminants in this media. Therefore we can foresee that contamination of the alluvial aquifer will continue to increase in time as concentrations of the diverse contaminants increase exponentially as the distance to the tailing dam decreases. This is due to the increase in volume of metallurgical waste accumulated in the tailing dam, which induces an increase in the recharge as infiltration gradient varies, while the climatic conditions favour infiltration of meteoric water.

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CHAPTER 10

Integrated international groundwater management: The Euro-region Praděd example

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ABSTRACT: In Europe, border areas usually belong to the most problematic regions; a border must not be considered as a line beyond which the local or regional authorities of one country are not interested in sources of negative influence on the area and the population of the neighbouring country (with regard to agriculture, management of ground, surface and sewage water, storage of waste or inadequate interventions on the landscape). Solutions to the problems of individual border areas are possible only if the legislative, natural, industrial, agricultural and historical backgrounds and also the human resources of the countries involved are mutually respected and if the needs and interrelations of a particular region are approached globally. To effectively manage trans-boundary waters and protect them against pollution, the governments of bordering states must agree upon common rules and actions. International cooperation between neighbouring countries is a privileged field that can contribute to sustainability, conservation and improvement of the whole environment, not only in Europe.

Motto: Where a better understanding of water quality and environmental awareness exists, there is the opportunity to ensure and provide a better quality of water to the population, which by all means should be achieved not later than the year 2010 (EU directive, 1997).

A trans-boundary issue may arise when a water system, due to its geographical location or political conditions, falls partly or totally under the jurisdiction of two or more countries. International cooperation is a basic instrument, when the catchment straddles two or more countries, in order to prevent, control and reduce transboundary pollution, and to ensure a sustainable water management for the conservation of water resources, with a view towards sustainable development (Nunes Correia, 1998).

The terms mentioned above ('trans-boundary issue', 'international cooperation') appear more and more often among the live issues quoted in the context of the problems of the water management and sustainable development across Europe. The theory of international cooperation in the area of utilization and management of trans-boundary water is modern and continually updated. However, its practise is poor. It is difficult to find a general model for trans-boundary management of water, whose conclusions would be applicable for all (or at least the majority) states in which the problems of trans-boundary management of water occur.

Management of trans-boundary waters (both surface and groundwater) is very complicated since there is not one government to manage international waters and neighbouring countries may have different languages, historical roots, cultures, institutional structures

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and water management legislation and rules. The number of agreements on trans-boundary waters in Europe is more than 150 and shows an increasing trend. The evolution of the European Community into a European Union (EU), as a consequence of the Maastricht Treaty, should imply a significant change in the relationships between all member states and trans-boundary issues should be dealt with in a more comprehensive way, which will certainly affect existing unilateral strategies and policies. The increasing number of countries applying for EU membership, namely from Central and Eastern Europe, will dramatically increase the need to better organize member state relationships with regard to the many environmental (including water management) problems.

In Central and Eastern Europe conditions for optimal trans-boundary management of waters are even more complicated, primarily at the boundaries of the so-called former Iron Curtain. These border areas are still the most inconsistent regions in Europe in terms of ecological balance, water management, land use, and social and economic structure, as well as agricultural and industrial development. Nearly fifty years of extremely diverging regional policy and a total lack of cooperation and interaction between the neighbouring states along the Iron Curtain have led to a significant discontinuity and destruction of former relationships and the competitive use of all types of common resources. Unfortunately, the situation is no brighter even at the boundaries between the once associated states. As an example of the practical utilization of trans-boundary management, I have chosen the boundary between two European associated states, the Czech Republic and Poland, i.e., the area limited approximately by the imaginary boundaries of Euro-region Praded.

The Czech-Polish borderland is generally characterized by some common features, but also some common problems. Previous development, the way economics functioned and, most of all, careless industrial production, have been the causes of substantial deterioration of the natural environment in these areas. In spite of the regional strategy for the protection of the environment and surface and groundwaters, with the support of environmental education, some problems persist in the Czech borderland.

In rural areas (and thus also in the Euro-region Praded) bad economic habits persist, as well as a disregard for soil and water conservation, the components of the territorial environmental stability system, water resources and for nature generally. It is not rare for oil resources to be exploited in forests, regardless of whether producing wood and resulting in deforestation. This situation is complicated by a new trend in property ownership; the new owners of the land and homesteads in the villages now live in distant towns, entirely outside the locality. Inappropriate methods of waste treatment still prevail and uncontrolled waste sites are still appearing in the countryside.

The prime problems of the pilot area, Euro-region Praded are of quantitative and qualitative types (Boukalová, 2001):

QUANTITATIVE PROBLEMS

Wide deforestation and intensive agriculture and industry on the Czech side of the borders are causing erosion and consecutively a decrease in the land retention potential. During heavy rains, this land retention potential decreases on the Czech side, causing excessive inflow of water, especially to Polish lowland areas, causing them to be liable to flooding.

Figure 1. Floods in the village Holčovice, Euro-region Praděd, 1997.

QUALITATIVE PROBLEMS

The southwest–northeast direction of water flow of both surface and groundwater is characteristic of the area, i.e. from the Czech to the Polish side. To date there have been different amounts of contamination, especially of surface water, but partly also of groundwater, on the Czech side, which endangers the Polish tapping areas.

Pollutants in Czech territory include:

- illegal junkyards
- side dressing
- insufficient or non-existent sewerage systems for the discharge of waste water from towns on to the land.

I will describe in more detail an example of these problems, which is directly related to the necessity of implementing integrated international water management in Praded.

Currently, general interest in the frontier territory of Písečná-Mikulovice-Glucholazy-Nysa in Praded is focused on the assurance of an appropriate quality of surface and groundwater, which is used as a primary source of drinking water by the inhabitants. While villages Písečná (CZ) and Mikulovice (CZ) are supplied mainly by groundwater from a Quaternary aquifer, inhabitants of Glucholazy (PL) (16,000 P.E.) and Nysa (PL) (55,000 P.E.) are supplied by an stream of the River Bělá, which flows through the villages of Písečná and Mikulovice to the Polish territory near the village Glucholazy.

The basic clean-up of the River Bělá began within the framework of the project, CREDO PHARE 1999 – Clean-up of the Bělá river. This EU project instigated the building of a sewerage system in the centre of Mikulovice and the continuous monitoring of the quality of the surface and groundwater flowing from the Czech Republic to Poland. Nevertheless, the avoidance of contamination leakage in Mikulovice does not ensure the appropriate water quality from the River Bělá because on the Czech side of the border there is also pollution from the Písečná-Mikulovice area, from which water drains uncontrolled into the river. This pollution creates the permanent potential danger of the breakdown of the supply of clean drinking water to the village of Glucholazy and the town of Nysa (concrete data may be determined only after repeated monitoring of the water quality in proposed sampling profiles has been performed, depending on the climatic, morphologic and hydrogeologic parameters of the area).

The existing management of trans-boundary water in the area is also complicated by the incompatibility of Czech and Polish legislation on water discharge from the waste water treatment plant to the watercourse and consequent intake of surface water for various purposes. After the floods of 1997, a waste water treatment plant was built in the village of Mikulovice (approximately 800 m from the border with Poland) regardless of Polish legislation, which states that drinking water may be collected from the river only at a minimum distance eight hours downstream from the treatment plant. However, on the River Bělá, a water supply draw point for the village Glucholazy is in the vicinity of the state border in contravention of the time stipulation. It would have been so simple to have avoided this problem from the beginning by considering the differences in the Czech and Polish legislation and, in order to comply with the required eight-hour delay, locating the treatment plant outlet 1,700 m further upstream.

The present circumstances mean that the village Glucholazy will, under no circumstances, agree to supply its inhabitants with drinking water from the River Bělá and will be obliged to search for a different solution. The easiest course of action is to establish a new groundwater withdrawal site. However, the establishment of a new Polish water withdrawal spring to supply drinking water to Glucholazy, which would utilize the until-now high quality groundwater from a deeper aquifer, is connected to another, even more complicated, conflict of interest. According to current Polish (and Czech) legislation, every water withdrawal point must have an appropriate water protection zone with partial/total restriction of agricultural and industrial activity. Polish hydrogeologists have proposed that the only potential water withdrawal point on the Polish side that meets these requirements and could be utilized for the purposes of Glucholazy and the surrounding area is in close proximity of the state border with the Czech Republic. This location for the potential new water withdrawal spring results in the necessity to determine an external protection zone in Czech territory as far as the town Jeseník, which would completely freeze present industrial production in the Czech part of Praded and inhibit further development of the region around Jeseníky.

HOW TO SOLVE THE EXISTING PROBLEM?

It is clear that in determining some basic rules for Czech-Polish cross-border water management, the conflict between Czech and Polish interests, which is growing from the need to manage water for the benefit of ordinary inhabitants in neighbouring countries, should not be allowed to escalate. Therefore it is necessary to discuss and enforce the following steps:

- to begin a programme of integrated international water managment in the region
- to solve the situation in such a way as to avoid the deepening of conflict on either side of the border (i.e. accelerated building of the remaining part of the sewerage system on the Czech side and provision of an alternative drinking water supply for Glucholazy)
- to establish close Czech-Polish cooperation and local action groups, which would solve potential future problems together.

RESULTS AND DISCUSSION

Start up of integrated international water management in the Euro-region Praded.

Initiation of the triennial scientific EU project, Integrated water management of trans-boundary catchments (TRANSCAT), where one of the pilot localities is Euro-region Praded (see later section).

Solution for the existing conflict

- preparation of the Instrument for Structural Policies for Pre-Accession (ISPA) project oriented to the water supply in the Jeseník area
- proposal for an alternative drinking water supply from the Czech side of the border for Glucholazy that can be used in case it is impossible to use water from the River Bělá (due to extreme hydrological phenomenon, industrial accidents, etc.)
- replacement of the waste water treatment plant outlet (i.e. moving it approximately 1,700 m upstream)
- the start of long-term monitoring of surface and groundwater quality in the area. It is proposed that a 20-month programme of monitoring surface and groundwater once a month is performed in both Czech and Polish territories. Monitoring should include:
  - groundwater in selected existing drill holes and wells in both territories (10 monitoring wells.)
  - groundwater of the River Bělá at five existing stable weirs and at the outlet from the waste water treatment plant
  - selected physical, chemical and microbiological parameters, within the meaning of the standard ČSN 75 7111 for drinking water, with the following minimum scope: suspended solids, BOD5, COD-Cr, N-total, P-total, pH, Eh, hardness, colour, odour, occasionally also reduced chemical water analysis (complete chemical and physical water analysis, respectively), non-polar extractives, biology plus bacteriology
- inventory and mapping of the potential sources of contamination of surface and groundwater.

Czech-Polish cooperation

The example of the River Bělá (and accordingly the Rivers Opava, Odra, and Ostravice) in this Czech-Polish borderland is characteristic of the conflicts of interest in trans-boundary water management, in that water is not protected or monitored in one state and is being used to supply drinking water in the neighbouring state. In Praded the problems of managing the utilization of trans-boundary streams are complicated by the problem of pollution of groundwater in the trans-boundary aquifer.

In the Czech-Polish borderland, one of the ecological problems is the fact that the two states do not enforce the same methods for monitoring pollution or apply the same preventative measures. A change in this situation can be expected in one to two years when a new directive, EC 2000/60/ES (the New Water Framework Directive), should be fully implemented. The directive offers a framework for community involvement in water policy (by the end of 2003, the directive should have been adopted into the national systems of law, including the determination of areas of catchments and competent authorities). The main aim of the directive (followed by new complementary laws, relating to water, in the Czech Republic) is to preserve water resources in the whole of Europe, at the same time ensuring the viability of a range of the uses, which will not endanger the future and sustainable utilization of water resources.

It is only thanks to the existence of the EC 2000/60/ES that it is now possible to prepare some rules for the management of trans-boundary water along the Czech-Polish border, complemented by an authorized model for a decision support system (DSS) for users. Moreover in the Czech Republic, development of the DSS for the management of trans-boundary water is facilitated by a trend in state policy towards environmental issues and the signing of the Aarhus Convention on Access to Information, Public Participation in Decision-making and Access to Justice in Environmental Matters. Enactment of these new regulations creates conditions for public participation in decision making on commitments and activities affecting water and the environment.

CASE STUDY: THE TRANS-BOUNDARY AQUIFER IN PRADED

What is a Euro-region?

Euro-regions are territorial entities joining together individual borderland areas of various states in Europe. These regions work voluntarily and their cooperation is either organized or unorganized. They have the advantage that towns and villages on both sides of the border can solve their problems together. This cooperation helps overcome the problems faced by towns and villages located in borderland areas and thus improves the living conditions of local residents. A Euro-region is most commonly formed by a voluntary union of villages, towns and counties with similar interests. Their cooperation is defined in the European framework contract on cooperation across boundaries between territorial units and authorities, enacted by the Council of Europe in 1980.

The basic aim of Euro-regions is to assist all activities at regional level, leading to involvement of the concerned countries in the European Union. In 1998 it was established a Secretariat of the Euroregion Praded (right after the devastative floods in 1997), which should, apart from other political and cultural activities, deal with the common land and water management of the area and make sure that all legislative, historical, ecological and human aspects of the area development are respected and all possible conflicts of interest of the Czech-Polish border area will be solved. The Euroregion is a voluntary association of the Czech and Polish societies and associations of towns and villages, includes approximately 60 municipalities on the Czech side and 13 on the Polish side. The main aims of Euro-regions are:

- cooperation on issues of territorial planning and arrangement
- preservation of the environment and amelioration of any negative impacts
- common access to the land and water management
- increase in living standard of the inhabitants
- advancement of cooperation in fighting against fire and natural disasters such as floods
- integration of the needs of border inhabitants, tourism and culture
- improvement of interpersonal relations and cooperation in social and humanitarian areas
- development of a public complaints procedure across national boundaries.

Currently the working structure of Euro-regions is formed on the basis of international contracts on cooperation, either for one-time activities, prepared for a specific purpose (e.g. the creation of trans-boundary bicycle paths), or for strategic cooperation for the general development of a region, as it is in Euro-region Praded.

Praded is a voluntary association of Czech and Polish towns and villages located in the districts of Bruntál and Jeseník in the Czech Republic and in the southern part of Silesia (district Opole) in the Polish Republic. The Euro-region was established by a Czech-Polish contract signed on 2 July, 1997. The Czech part of the regions was registered as an association of interested juridical persons by the district authority at Bruntál on 9 October, 1998. The total area of the region is 3,300 km^2, the Czech part being 1,600 km^2 and the Polish comprising 1,700 km^2.

What is a trans-boundary aquifer?

The typical trans-boundary aquifer includes a natural subsurface groundwater flow intersected by an international boundary such that water transfers from one side of the boundary to the other. In many cases (as in the Praded case study) the aquifer may receive the majority of its recharge on one side of the boundary (Czech in this case), whereas most of its discharge occurs on the other side (Poland).

The trans-boundary aquifer area of Praded includes natural conservation areas in the Czech Republic (the protected land reserve of the Jeseníky Mountains) as well as in Poland (the protected park of the Opawskie Mountains), which represent some of the most attractive and valuable reserves in either country. However, this area is also characterized by the impact of quite inappropriate human intervention into the hydrogeological regime, dramatic deforestation, the inappropriate planting of spruce monocultures, extensive agriculture and unregistered waste sites.

From the geomorphological view, the area belongs to westernmost portion of Eastern Sudeten. From the climatic point of view, the area belongs into the slightly warm region, in the North into the slightly warm hilly and flat district, moist with mild winters, in the

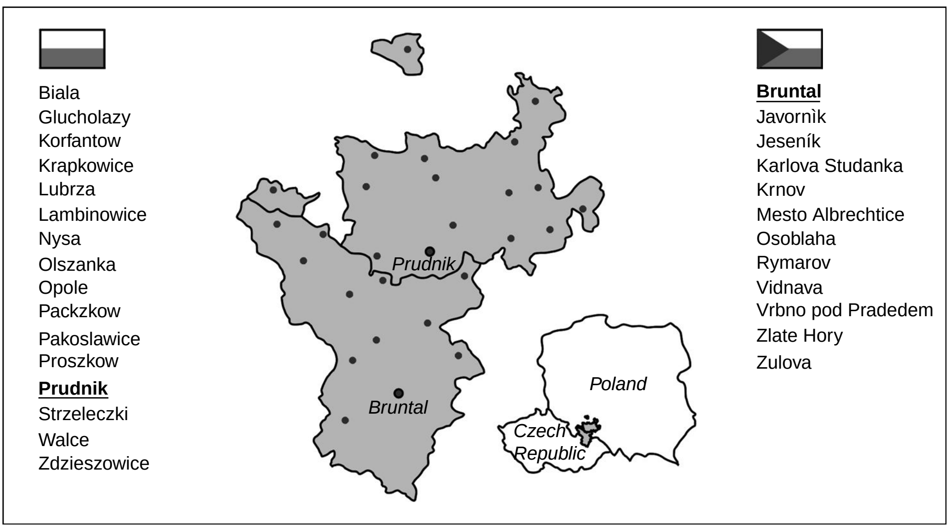

Figure 2. Euro-region Praděd.

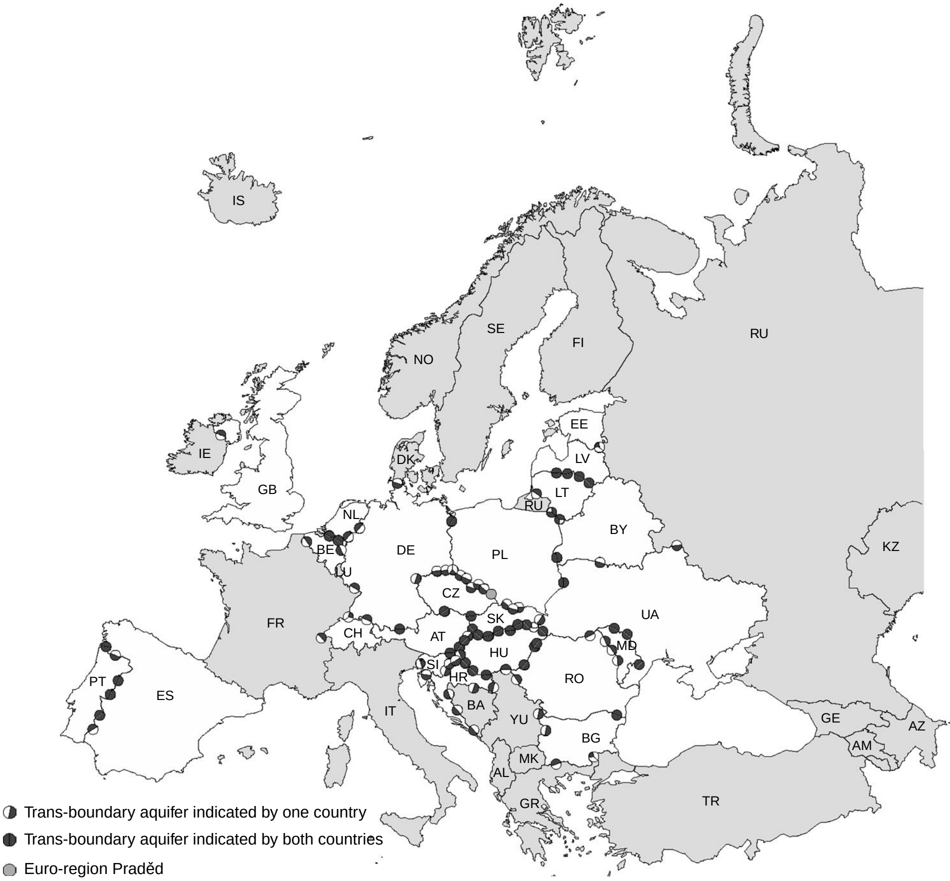

Figure 3. UNECE Survey of European trans-boundary aquifers from Puri et al., 2001.

South into the slightly warm, moist highland district. The number of freezing days in the northern part of the area is 100–120; the number increases up to 180 in the southern direction. The number of days with snow coverage in the northern section of the area is 50–60; in the southern direction it increases up to 120. The dominant wind directions are the southern and southwestern; northern, northeastern and western winds occur approximately as frequently. The least frequent are northwestern and southeastern winds. As concerns the climate, there are three climatic types in the area. In the northern and northeastern portion of the area in question, there is the clime of a hilly region; in the southern portion, there is the clime of rocky regions and in the southwestern portion, there is the clime of highlands. In the central portion of Rychlebské hory, there is also the clime of rocky region. The land use in the Polish part of Praded is mostly agricultural, but there are also grassy growths and wetlands. On the Czech side, there are two types of landscape – forest with pine trees and a significant predominance of grassy growths.

The modern development of the fluvial networks in the area occurred only after the receding of a continental glacier, when a gradual uplift of the entire area occurred and the riverbeds carved into the bedrock. The area drains northward to the Baltic Sea and is part of the River Odra basin. There are two hydrogeological regions: crystalline complex with permeability given by fissures and porously permeable glacial-fluvial sediments (sands, gravel sands, gravels and clays).

This part of Praded was selected as a pilot site primarily because the question of common management of water in the area is quite timely and has even, perhaps, reached a critical stage. There is a significant need to introduce and adopt state-of-the-art information access tools at end-user sites. Focus on the innovative use and integration of soft computing, statistical, simulation and optimization methodologies and related solutions is essential for the coherent and transparent development of this trans-boundary region. Generally in trans-boundary areas there is a need to restrict all existing and potential clashes of interest caused by: the lack of information about the legislature of neighbouring states; uncoordinated activities during changes in the infrastructure; not respecting the cultural and ecological objectives of neighbouring states, often based on tradition and historical roots; and the total unavailability of mechanisms for rapid response and exchange of the important (compatible) information necessary for complex regional evaluation. Scientific innovation in Praded could be demonstrated in a verified structure of information that can be retrieved from all information sources nationwide.

Solution for the trans-boundary problems: integrated land and water management

The best way to provide trans-boundary management is to design a system that will integrate data from different sources. The main part of the system should be a database of the legislative rules of the frontier countries, in the context of the Geographical Information System (GIS) levels used and the problems of the end users. The main objective of appropriate trans-boundary management should be the analysis of existing treaties, agreements and cooperation documents between international institutions, as related to water resources issues, in order to find patterns of behaviour relevant to the analysis of the problems and proposed solutions. The aim should be to identify the advantages and disadvantages of each course of action. A very important part of trans-boundary water management will be the institutional framework that is set up to implement the decisions and commitments of the relevant countries.

In practice, the problems of trans-boundary management of water has been solved by only a very limited number of international projects. In Euro-region Praded, a series of these projects are being performed (or have been performed in the past). The following projects are among the most important ones for the Czech-Polish area.

The **EC project, Coherent International Environmental and Groundwater Monitoring and Management** was financed within the 5th Framework Programme and closed in October, 2001. The main objective of the project was to create working groups of end users in the area and to collect the data necessary for water and land management for future use and integration.

The NATO project, **Trans-boundary Environmental Monitoring and Management with High Resolution Satellite Remote Sensing** (TEAMSAT) was financed by a collaborative linkage grant from the Cooperative Science and Technology Sub-Programme. The main objective was the application of long-distance land monitoring and Geographical Information System (GIS) to cross-border management.

Figure 4. Brantice, a field-worker evaluating the water quality during the regular extraction of drinking water in Praděd.

The main objectives of the **PHARE-CREDO project** on infrastructural changes in the Mikulovice-Glucholazy area are the construction of a sewerage system, development of a groundwater quality monitoring system and data collection.

The UNESCO, FAO, UNECE and IAH framework programme, **Internationally Shared (Trans-boundary) Aquifer Resources Management (ISARM)** had the purpose of summarizing the current understanding of trans-boundary aquifers, demonstrating their significance in terms of water resource management and highlighting the fact that as yet there is very little experience in the approaches needed for shared water management. Euro-region Praded was studied under the ISRAM programme as the one of six of the world's shared aquifers.

The main goal of the EC energy, environment and sustainable development project, **Integrated Water Management of Trans-boundary Catchments** (TRANSCAT) (Boukalová, 2002) is to create a comprehensive operational and integrated decision support system (DSS) for optimal water management in the catchments of borderland regions, within the context of the implementation of the EU Water Framework Directive. Another objective is the establishment of a European platform for initiating and promoting international cooperation and networking that will allow a more detailed insight into trans-boundary water, environment and related socio-economic problems. This will be achieved by integrating the TRANSCAT research with that from previous EU research for framework programme 5 and connecting the project with EU institutions involved in the process of implementing the Water Framework Directive (WFD), with specific reference to the problems of pre-accession countries.

During work on the project in Euro-region Praded, several general basic rules were deduced, which can be used to start to establish international cooperation within the management of trans-boundary surface and groundwater, as follows.

1. Integrated management across water uses and jurisdictional boundaries is the key to sustainability.
2. Water development and management must move from a project-by-project focus to systematic planning and systems management.
3. Technical expertise is necessary, but not sufficient, for achieving integrated water management. The political agreements and good prepared End-Users background are necessary.
4. For the successful management of integrated trans-boundary water, it is convenient to use some of the existing or developing models and systems (e.g. DSS). In this context, it should be realized that data are rarely neutral. Data collection itself is driven by certain values and is ultimately negotiable. Therefore, it is convenient to use such models as are economical and readily available to end users.
5. Precise definition of the Awareness creation and End-Users background in the trans-boundary region is under preparation via following actions (should be finalised in the spring of the year 2005):
   - analysis of the existing legal and institutional framework as related to trans-boundary water resource systems
   - analysis of the historical, political and economic roles of trans-boundary regions/countries
   - identification of the motives behind conflicts and problems that demand various forms of international collaboration
   - identification of the relevant existing and potential conflicts in trans-boundary water areas in all categories
   - identification of new concepts, proposals and strategies for development within the trans-boundary framework
   - identification of the policy factors that influence a country's position on international catchment
   - identification of the most important interest groups, activation of their involvement and setting up local regional steering committees and observer groups
   - provision of information to the public by organizing information events
   - preparation of acceptance and support measures for the performance of local work for test implementation
   - presentation and discussion of possible scenarios and strategies for recommendations for optimal trans-boundary water management.

End users' background

The period of communist rule left Poland and Czech, like many other countries of this part of Europe, with serious environmental and water resource problems. Many of present problems have roots in the past: In some villages, there is up to 30% rate of unemployment, people have no resources nor motives to make business, what results in their low economic standard. Young people stay in towns (population in the villages grows older, required capacities for the schools and nursery gardens are missing). Its development is also slowed by a bad infrastructure. Cooperation with the Polish side is also not on the best level. In Poland, the regional transport to the Czech Republic is reduced (reduction of tourist business). There is also a problem of low donations from EU. E.g. from the programme INTRREG III, only half of amount of money might be invested into the whole Czech-Polish border, in contrast with the Czech-German border, where about twice much might be invested. The cross-border economic and social co-operation is not very developed. Even the existence of the Euroregion Praded, which includes the area here in question, and is geared towards the activation of the cross-border collaboration in various domains, helps only to a limited extent. For enabling that the good End-Users background and cooperation will be established on the transboundary area, it is necessary to thoroughly solve (except the professional scientific problems of the water management) also the social economic question of its application: the bullet list follows, starting with: – Possible concurrence of interests (the interdisciplinary, inter-sector, cross-border and competence ones), etc.

- to ensure that a good end users' background will be established for the trans-boundary area, it is necessary to thoroughly solve the socio-economic problems arising from the application of water management solutions
- agreement on interdisciplinary, inter-sector, cross-border and competence interests
- creation of a structure for decision making in the areas related to the competencies of individual ministries in relation to the problems of water treatment (i.e. water management, environment protection, regional development of the countryside, land management and agricultural/forestry/fisheries use of the countryside)
- in developing a good end users' approach and water management strategy the present mechanisms of state communication structures should be considered and utilized (a proposal for new mechanisms for communication would be counter-productive and may cause incompatibility between the model and the demands of the end users, thus degrading the scientific research
- consideration and accentuation of the compatibility of economy management in the selected case study areas, with access to the EU (in the first place of the WFD)
- the solving of concrete problems in the selected localities, with consideration of local interests (these may be in conflict with regional ones), thorough analysis of the problems, testing of possible solutions and definition of a strategy for the optimal solution.

Globally, the integrated international groundwater management should be connected to the needs of real end users and stakeholders, backed by the following new laws:

- Drinking Water Directive
- Environmental Impact Assessment Directive
- Water Framework Directive.

It should also be in total agreement with WFD goals of 'getting Europe's water clearer and getting the citizen involved'.

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CHAPTER 11

Groundwater recharge estimations from studies of the unsaturated zone

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ABSTRACT: This paper presents two independent methodological approaches for estimating groundwater recharge from data collected in an experimental plot located on a plain in Argentina. The first method is a direct application of the Darcy equation at a flow plane 120 cm deep, whereas the second method took into account a water balance based on groundwater-level data using the Visual Balan program. A time period of 711 days was used, after which both methods yielded a recharge of 11 per cent (expressed as a percentage of the rainfall in the same period). The first method concentrated the recharge events in autumn, whereas the Visual Balan was able to reproduce smaller recharge events probably related to the existence of preferential flow paths.

INTRODUCTION

Supplementary irrigation, which relies almost entirely on groundwater resources, has steadily grown in many regions of the Argentine humid pampas in the last years. Although no reliable records of the number of wells drilled or the individual discharges exist, it is evident that there has been an increase in the use of groundwater resources. For instance in the study area, located in the centre of Buenos Aires Province, individual wells may yield more than 120 m^3/h, yet their operation is not ruled by the provincial authorities. Given such a situation, appropriate knowledge of the recharge to the regional aquifers is needed in order to help shape future policies regarding water resources management.

LOCATION

The study area is located in the Azul River basin, Buenos Aires Province, Argentina. It correspond is extensive, flat land with regional surface slopes in the order of 0.5–0.8 per cent. According to the Thornthwaite classification, the climate is sub-humid to humid, with medium range of temperatures and little or zero deficit of water. The mean annual precipitation is around 900 mm, based on the maximum values recorded in March and the minimum in August. The mean annual temperature is 14°C, with a mean maximum of 21.5°C in January and a mean minimum of 7.2°C in July.

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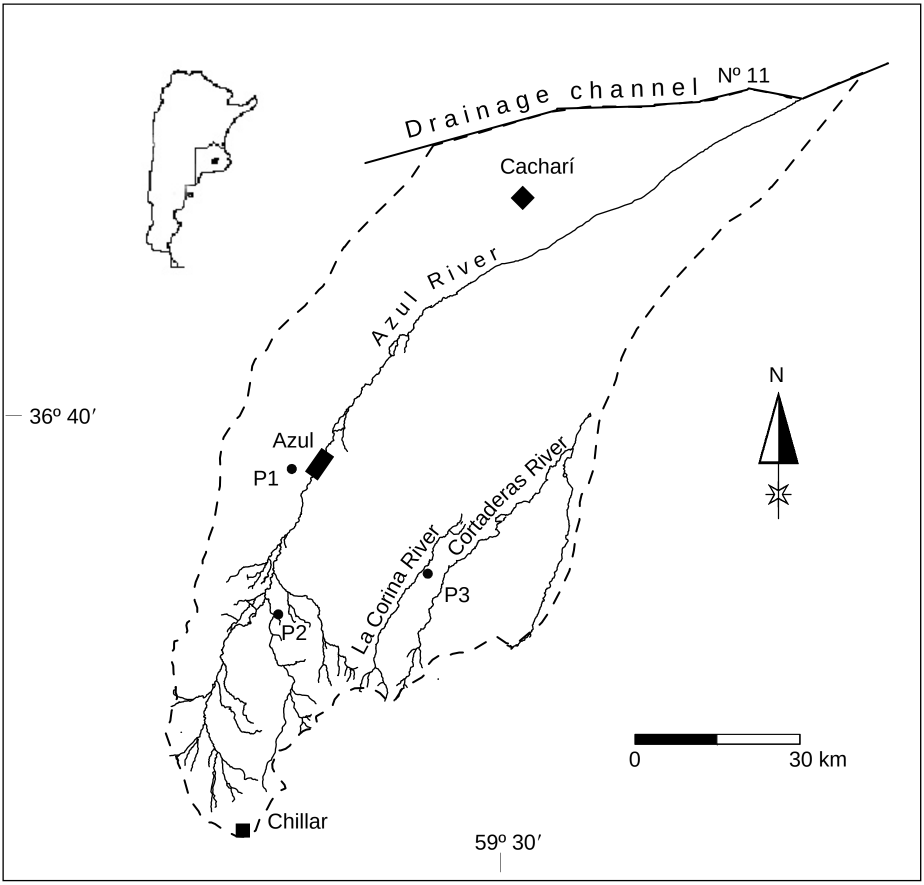

Figure 1. Location of the study area and experimental plots (P1, P2 and P3).

The upper basin consists of a low hilly area and the lower basin is a large plain. The hilly area is physiographically connected to the plain by pediments (middle basin), where three experimental plots were set up to assess the characteristics of the unsaturated zone (Figure 1). It should be mentioned that Varni et al. (1999) have found that such an area contributes most of the groundwater recharge due to the properties of the existing soil types.

Data from one of the experimental plots (P1 in Figure 1) were used to estimate groundwater recharge by means of different methodologies. Preliminary analyses indicated that such a comparison would be successful at any of the plots and the reason for selecting P1 was because of its larger data sequence. Argic soils (notably Argiudolls) cover the study area. At P1, such a soil is about 1 m thick, with an upper portion made up of clay loam, two clay horizons (Bt1 and Bt2) between depths of 18–66 cm, whereas the deepest horizon (Ck) is silt loam. Discontinuous caliche layers are common. Roots are abundant in the upper 20 cm and penetrate as far down as 70 cm in depth.

These soils are primarily used for agriculture, with remarkable harvest yields, and pastures are often grown for cattle grazing.

Groundwater levels are shallow, in general no deeper than 3 m, and rise to less than 1 m below the surface when the rainfall events are important.

METHODOLOGY

Two independent methods were employed for calculating the recharge and comparing their results. First, the Darcy equation for unsaturated conditions was applied (Weinzettel & Usunoff, 2001), by assuming that the recharge is represented by all water fluxes that cross a plane arbitrarily set at a depth of 120 cm (far down from the influence of roots). Second, a water balance for the unsaturated and saturated zones was carried out by considering the Visual Balan program (Samper et al., 1999), which took into account the unsaturated zone parameters measured at P1. Both methods were applied and compared for a period of 711 days, starting on 21 October, 1998 and ending on 30 September, 2000.

It should be noted that, in accordance with Sophocleus (1991), this paper refers to recharge as effective recharge, that is water that percolates into the lower limits of the vadose zone, reaches the water table and causes a measurable rise in the water table.

Determination of hydrodynamic characteristics of the unsaturated zone at P1 was based on five digital tensiometers installed at various depths (15, 30, 60, 90 and 150 cm), each with access tube for a TDR probe, class-A evaporation tank, pluviometer and phreatimeter. Daily readings were taken, except for soil moisture, the values of which were determined weekly. The plot is covered by short natural grass like the surrounding prairies.

FLOW IN THE UNSATURATED ZONE

Darcian flow in the unsaturated zone was estimated at a plane located 120 cm below the surface where the effect of evapotranspiration can be disregarded.

The Darcy equation for unsaturated conditions may be written as follows:

$$q = -K(\theta) \cdot \nabla H \tag{1}$$

where q is the flux that crosses the 120-cm deep plane, $K(\theta)$ is the hydraulic conductivity as a function of soil moisture at 120 cm depth and ∇H is the hydraulic gradient according to the daily readings of the 90 and 150-cm deep tensiometers. It was assumed that the tensiometers were located in positions that avoided the influence of the root zone and capillary fringe (see Weinzettel & Usunoff, 2001 for greater details). The method has been applied in many studies in arid, semi-arid and humid conditions (Scanlon et al., 2002).

In equation (1), $K(\theta)$ is of fundamental importance for calculating the unsaturated flow. In this case, the $K(\theta)$ function came from an internal drainage test (Hillel et al., 1972; Villagra, 1992). Such a test implied complete saturation of the soil profile and covering the experimental plot with a plastic liner to eliminate evapotranspiration during the 30-day test. Several readings of tension and water content were taken, which allowed the assessment of the $K(\theta)$ function for a depth of 120 cm (Equation 2 and Figure 2). The saturated hydraulic conductivity, for the maximum water content measured during the test, was 66.2 mm/day. Notice the rather low values of the local hydraulic conductivity in view of the restricted range of variation of the soil water content for the time span of the test. Values of hydraulic conductivity are sensitive to small changes in water content.

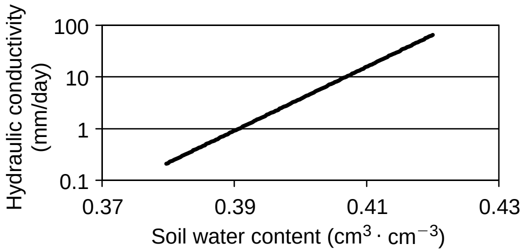

Figure 2. Relationship between the soil water content and the hydraulic conductivity at a depth of 120 cm.

Nielsen et al. (1973) pointed out that characteristically the hydraulic conductivity value decreases by orders of magnitude for only a small decrease in water content.

$$K(\theta) = 9 \times 10^{-25}\, exp(141{,}81 \times \theta) \tag{2}$$

The water content at a depth of 120 cm was measured every 7–10 days. An average value of θ was assumed to be valid for the periods between successive readings was and used to estimate $K(\theta)$ according to Equation 2. The integration of all fluxes calculated by means of Equation 1 gave the recharge for the period (Figure 3). It has to be mentioned that hydraulic gradients indicated that in some cases flow at 120 cm depth had to be upwards, although in practice it was deemed that this was not the case due to low soil water contents.

SOIL WATER AND GROUNDWATER BALANCE MODEL

In order to obtain estimates of recharge to the aquifer that may be compared to the method above, the Visual Balan program (Samper et al., 1999) was employed. Such a program calculates the water flows in the unsaturated and saturated zones, based on measured groundwater level fluctuations, and computes aquifer recharge. To do so, it carries out water balances in the upper zone of the profile (root zone), the vadose zone and the upper portion of the aquifer. Although the program is able to perform automatic calibration of parameters, this option was not used because many reliable parameters were available. Thus, the calibration was done manually, particularly on those parameters of an empirical nature. The code was set up to compute the water balance for a working area unit of 100×100 m. Measured groundwater levels were used to fit some of the parameters of the model.

The following sections describe the sources of the input data.

Evapotranspiration

Potential evapotranspiration data were fed to the program using estimates coming from an evaporation tank installed on the plot. Notwithstanding the difference between pan-evaporation and the evapotranspiration of cropped surfaces, the use of pans to predict ET_o for periods of ten days or longer may be warranted (Allen et al., 1998; Jensen, 1973).

Estimates of evapotranspiration from pan-evaporation make use of a coefficient that was obtained, following Doorenbos & Pruitt (1976), on the basis of wind speed, relative

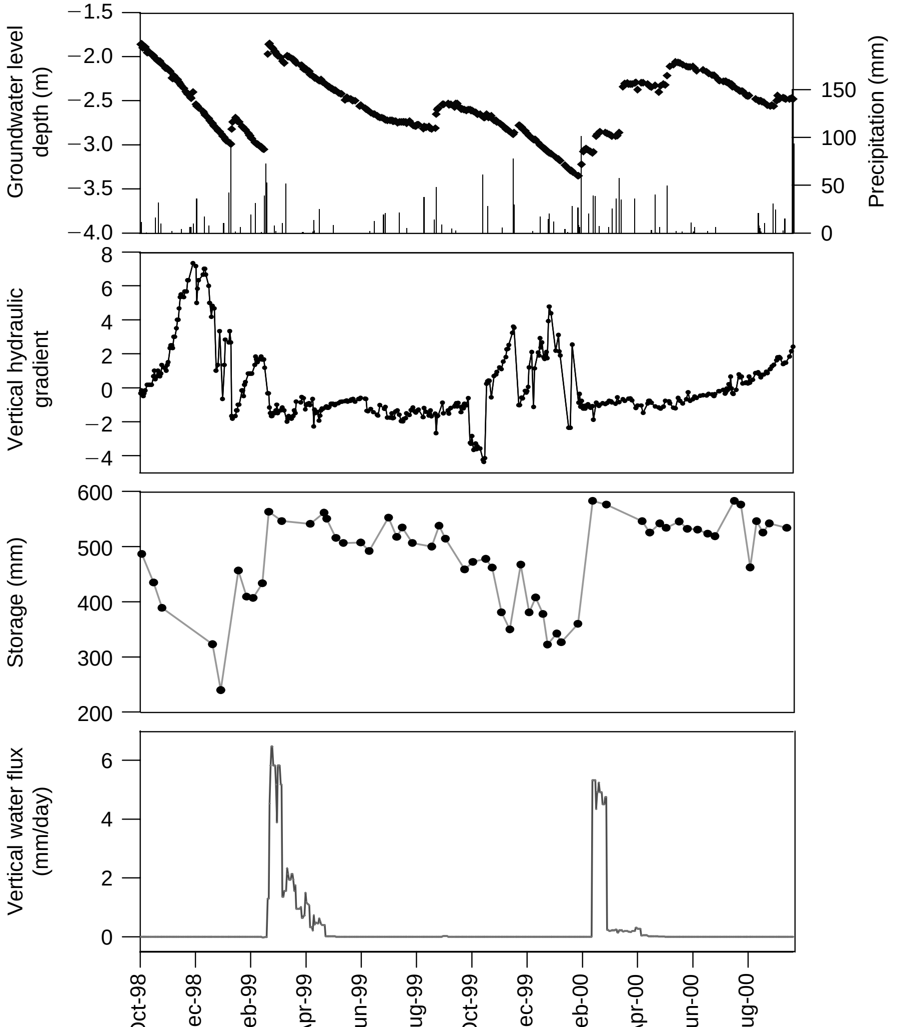

Figure 3. Recharge to the aquifer according to flows at 120 cm depth (Weinzettel & Usunoff, 2001).

humidity and distance to relevant vegetation masses. According to the daily climatic conditions in the region, such a coefficient ranged from 0.65 to 0.75. Given those values, potential evapotranspiration for the period turned out to be 1.900 mm.

In order to estimate real evapotranspiration, the Penman-Grindley option available in the Visual Balan program was selected.

Soil

Data such as thickness, field capacity, permanent wilting point and total porosity were taken from the measurements from the plot, assuming an average of the retention curves available for each soil depth (Weinzettel & Usunoff, 1999). Thus, the total porosity was

50 per cent, the field capacity 40 per cent, the permanent wilting point 25 per cent and soil depth was 0.9 m, which gives a water storage of 135 mm.

The total saturated hydraulic conductivity (K_s) was obtained as an average of the saturated hydraulic conductivity for each soil horizon weighted by the respective thickness (Jury et al., 1991). Estimated in this way, K was 25.2 mm/day.

Earlier studies (Weinzettel & Usunoff, 1998) reported the existence of by-pass flow (Beven & Germann, 1982), whose source is macroporosity and/or preferential flow paths, so that water enters the aquifer by-passing the soil matrix. Therefore, a coefficient that activates infiltration through preferential flow paths was applied. This is a very convenient to simulate the real situation in those cases that water ponding at the surface induces rapid recharge.

Aquifer and vadose zone

Both the aquifer and the unsaturated zone are made up of silty sediments (loess-like). Taking into account the water level response to selected rainfall events, a value of 7 per cent was assigned to the effective porosity. That figure is within the range of effective porosities for such sediments. Indeed, as explained by Sophocleous (1991), reliable effective storativity values for each recharge study site can be obtained by associating rises in water table with specific precipitation events and combining the recharge estimates from the soilwater balance analysis with consequent rises in water table. For the unsaturated zone and the aquifer, a value of 66 mm/day was assumed for the vertical hydraulic conductivity. The remaining parameters for the aquifer and the vadose zone were calibrated.

Model results

Visual Balan requires that the measured input data (groundwater levels) be given for 365-day periods. Therefore, data for 21 October, 1998 and 20 October, 2000 were used. In doing so, the simulation covered 20 days more than the first method, which could not be applied because the rise of the water level at the end of September 2000 saturated the deeper tensiometers.

Figure 4 depicts the groundwater levels computed by Visual Balan compared to the actual measurements. Overall, the fit is good as far as the shape and dynamics of the water level rise and decline. The left extreme in Figure 4 shows that the program simulates groundwater levels somewhat higher than the actual ones, whereas the opposite happens during the Spring 1999 season. There is an acceptable fit during 2000, with a slight tendency to overestimate the groundwater levels.

The mean square error (measured vs estimated values) turned out to be 0.16 m, which was acceptable at this stage of the research to compare results from the different methods. Figure 5 shows a linear function that relates measured and calculated groundwater levels with respect to the land surface. The largest differences occur when the groundwater levels are relatively deep (2.5–3 m below the surface), which may indicate that at such depths the unsaturated zone parameters and/or those of the aquifer are no longer constant.

If a two-year period is considered (Figure 4), the total precipitation is 2.093,6 mm and Visual Balan estimates 348 mm of recharge. For the same period, the total surface runoff

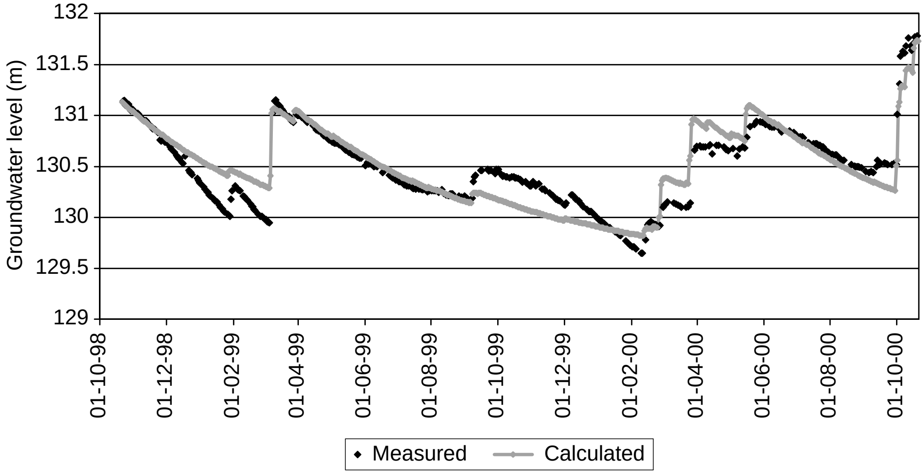

Figure 4. Measured and calculated groundwater levels at the experimental plot P1.

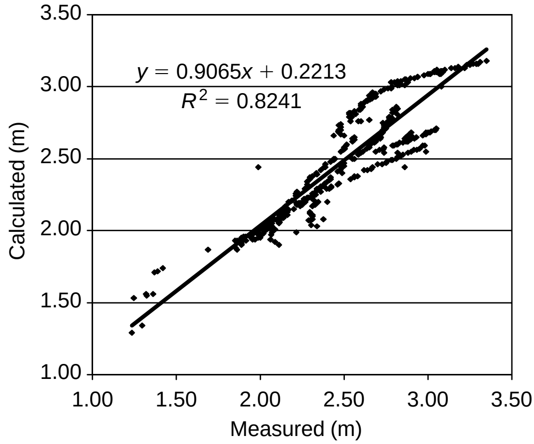

Figure 5. Fit between measured and computed groundwater levels.

according to Visual Balan was 25.1 mm, or 1.2 per cent (concentrated in a few days of March 1999 and March, May and October, 2000).

Surface runoff is a small portion of the total input by rainfall and its eventual occurrence coincides with the heaviest precipitation events, which produce aquifer recharge (autumn and spring seasons). Such a water excess, which Visual Balan computes as surface runoff, in reality leads to water ponding. It has not been observed, even during exceptionally intense precipitations, that water leaves the area as surface runoff. In the study area, all excess water (which would otherwise produce surface runoff) ends up being evaporated or infiltrated, which once again confirms that in large plains the vertical components of the water balance dominate over the horizontal ones.

DISCUSSION

The results obtained with the Darcian approach (details can be found in Weinzettel & Usunoff, 2001) are: groundwater recharge of 216.3 mm for the 711-day period (up to 30 September, 1999) for a total precipitation of 1936.3 mm. For the same period, Visual Balan rendered an estimate of 229.8 mm.

Table 1 summarizes the results of both methods for each hydrological year, as well as for the whole period. For the 711-day period, the recharge (given as a percentage of the rainfall) is similar: 11.8 per cent from Visual Balan and 11.1 per cent from the Darcy approach. However, some differences emerge when analysing the monthly totals (Figure 6). During the first year, the flow plane method (Darcy) led to flows greater than those computed by Visual Balan, whereas the opposite occurred from October 1999 on. The greatest differences were found in March and April 1999, and April 2000 (Figure 6).

In general, as expected, both methods associate larger recharges to the relevant rainfall events. Visual Balan, however, has detected a larger number of small recharges throughout the year. That might be attributed to the existence of preferential flow paths, the effect of which was taken into account in setting up the program. The flow plane method, on the other hand, cannot pick up by-pass flow because it is based solely on matrix flow.

In the long run (over the whole period), the overestimations and underestimations from both methods compensate, so that the amount of water recharged is quite similar.

The flow plane method computes the local recharge without consideration of the groundwater level fluctuations in the period. Visual Balan, on the other hand, solves a complex water balance by reference to the measured groundwater levels. The fluctuations of such levels may be influenced by lateral increased flows coming from the surrounding areas. It is deemed, however, that such flows are relatively unimportant with respect to the local evapotranspiration and infiltration phenomena.

Inasmuch as the Darcian approach relies heavily on the K(θ) function, such a relationship has to be carefully validated by new internal drainage tests, Van Genuchten-type models or other methodologies that may help reduce the inherent uncertainties.

Table 1. Groundwater recharge comparison.

		Visual Balan	Flow plane
October 1998–October 1999			
Precipitation (mm)	848.8		
Recharge (mm)		89.1	129.5
Recharge (% rain)		10.5	15.2
October 1999–September 2000			
Precipitation (mm)	1087.5		
Recharge (mm)		140.7	86.9
Recharge (% rain)		12.9	8.0
October 1998–September 2000			
Precipitation (mm)	1936.3		
Recharge (mm)		229.8	216.5
Recharge (% rain)		11.8	11.1

Shortening the frequency of field measurements is also recommended, particularly during the seasons of heavy rainfall.

It may also be advisable to apply both methods for longer periods to check whether the differences detected persist and, in that case, how relevant they are, in order to come up with acceptable values of groundwater recharge. The Darcian approach has is limited by the depths of the reference flow plane and the water level, both of which may be subject to the influence of the capillary fringe. As said before, the assumption was made that it could be considered as negligible for the time period selected, although it should be taken into account in those cases where rising groundwater levels approach the lower tensiometers.

The methodological comparison presented here reveals that either of them is valid for the study plot as long as the limitations of each application are acknowledged. In terms of relative error, the uncertainty related to recharge estimations cannot be below that attributed to the data and variables used along the methodological procedure (Samper, 1998).

Mention should be made of the results of the chloride ion mass balance for the period 1994–99, which indicated a mean recharge of 9.7 per cent of the annual rainfall (IHLLA, unpublished data, 2000). This method assumes that piston flow is the dominant mechanism of recharge. Although the cited figure is close to the recharge values presented here, it should be taken cautiously because ongoing studies (Weinzettel & Usunoff, 1999) indicate that local recharge has an important component of preferential flow. As a matter of fact, Weinzettel and Usunoff (1998) obtained samples of the chloride ion travelling through the unsaturated zone of the study area and found that chloride contents increase with depth, but it decrease in samples right above the water table, which was interpreted as the existence of preferential flow paths (Allison, 1994), disregarding the dilution effect of the lateral flow (Sharma & Craig, 1989).

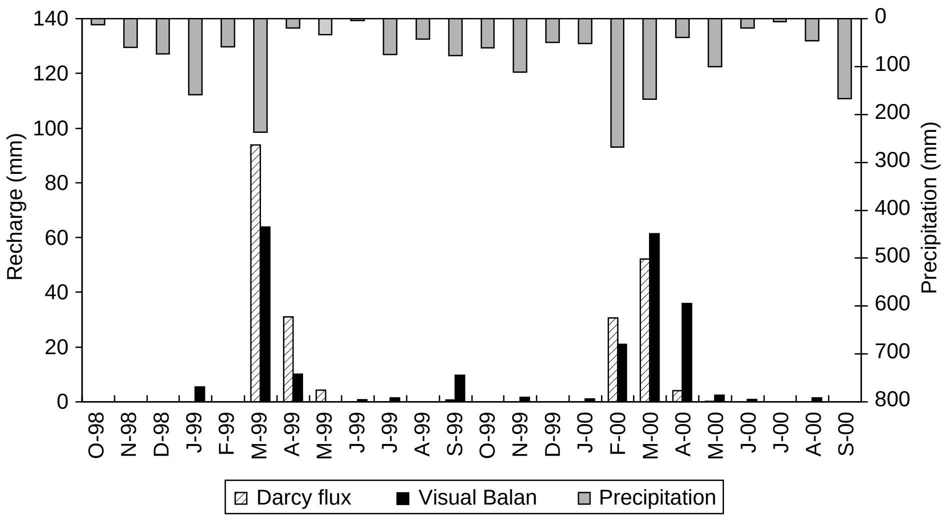

Figure 6. Comparison between the recharge according to Visual Balan and that obtained from Darcian flow in the unsaturated zone.

CONCLUSIONS

Two independent methods, which take into account the features of the local soils and the unsaturated zone, were applied and compared in order to estimate the recharge for a 711-day period. The recharge, measured as a percentage of the rainfall during the period, gave comparable results. Likewise, the distribution of recharge during the period followed a similar pattern.

Although the methods have limitations, it would appear that their well-considered use can render good results. The Darcian approach implies a great deal of uncertainty related to the validity of the $K(\theta)$ function, whereas Visual Balan performance is quite sensitive to some input parameters, which may not always be available or whose computation is not easily carried out. Nonetheless, their complementary use in the study area gave satisfactory results.

The availability of records for two years in the experimental plot was a major point in defining some of the parameters required by Visual Balan, which, in turn, was able to come up with an aquifer recharge function distributed along the simulated period.

The recharge values obtained are comparable to those coming from chloride ion mass balance, taking into account a much larger time period.

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CHAPTER 12

Geohydrology in plain areas: A conceptual model of a complex system, Los Saladillos, Santa Fe Province, Argentina

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ABSTRACT: This paper presents the synthesis of studies carried out to determine the conceptual model of a complex geohydrological system in plain areas located within Los Saladillos Basin, Santa Fe Province, Argentine Republic. This hydrogeological system shows a high degree of complexity, determined by the composition of the sedimentary column, the variability of the formational hydraulic parameters, the differential hydrochemical characteristics and the influences of present landforms in its shape. The definition of such a system requires a careful analysis of large amounts of information and the possibility of synthesizing the thematic aspects, excluding preconcepts based on the knowledge of the conventional hydrogeological system behaviour. By means of the convergences of geological, geomorphologic, hydrodynamic, hydro-stratigraphic and hydrochemical evidence a logical approach to the behaviour of the system was elaborated, on the basis of its being formed by two sub-systems: one of shallow fresh waters and another of deep salty waters, which are interrelated in Los Saladillos Basin. This interrelationship is controlled by an interface of salty fresh water that is dynamic and sensitive to the hydraulic head as a consequence of the hydrological situation in the River Paraná valley.

INTRODUCTION

The lower basin at Los Saladillos is in Santa Fe Province, Argentina and is located between 29° 00′ and 31° 40′ South and 59° 40′ to 60° 30′ West. Its surface is approximately 3,500 km^2 (Figure 1). The drainage network is composed of two main streams called Saladillo Dulce and Saladillo Amargo. At the centre of the lower basin they joint each other and become a single stream called Saladillo.

Large plains generally contain geohydrological systems subject to the surface morphological expressions and geohydrological systems completely independent of the

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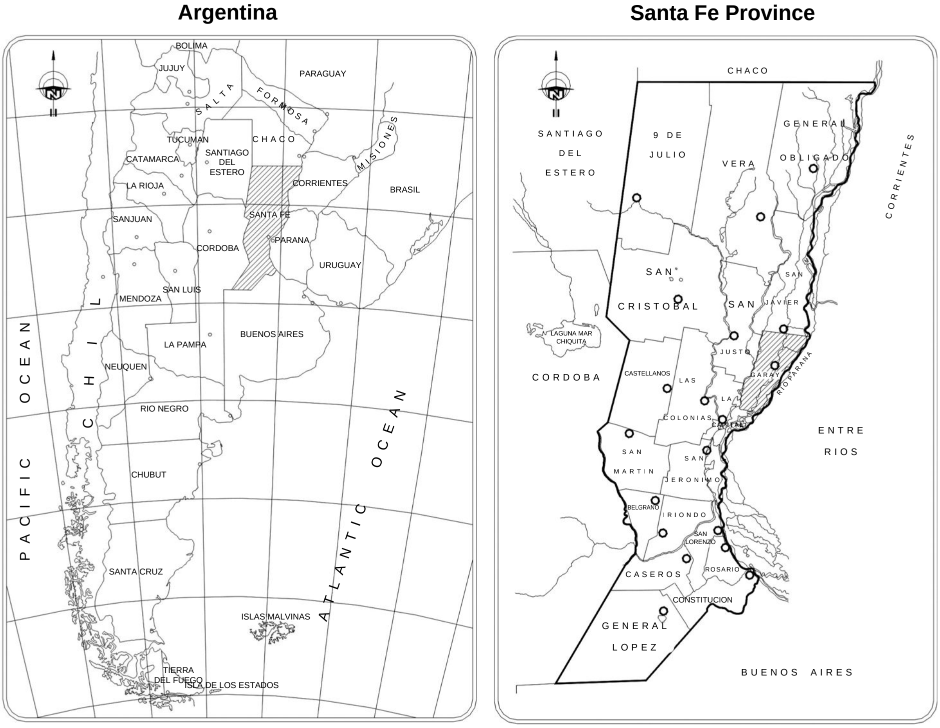

Figure 1. Location of study area.

local topography and climatology. Between such situations there is a great diversity of hydrogeological conditions and different interrelation levels affecting deep and shallow aquifer systems. This fact makes the conceptualization of the system behaviour difficult.

Situations such as the ones described above can be found on the Argentinean Chaco-Pampeana Plain, with very different hydrogeological characteristics from one region to another. Los Saladillos Basin is a representative area of the hydrogeology of these environments.

MATERIAL AND METHODS

Basic concepts of systems theory (Doménico, 1972) were used to support the integrated outlook that must be achieved when analysing the different components for understanding the hydrogeology behaviour in plain areas.

Basic geological, stratigraphical, structural, hydrometeorological, hydrological, hydrogeological and hydrochemical information was obtained and it was minutely analysed.

Diverse methodological procedures such as univariate and multivariate statistical analysis, geostatistics and those related to hydrogeology were used in the data processing, as well as in optimizing the results and making them compatible.

Stratigraphy

AGE	FORMATION	LITHOLOGY
HOLOCENE	Recent and Current Alluvial Deposits without differentation	Sands and clayed silts
UPPER PLEISTOCENE MEDIUM PLEISTOCENE	"Grupo Pampa"	Clays, silts and loess
LOWER PLEISTOCENE UPPER PLIOCENE	Ituzaingó ("Puelches")	Quartzose ocrate fluvial sands
UPPER MIOCENE	Paraná	Green clays and marine gray sands
LOWER MIOCENE EOCENE	Fray Bentos/Olivos	Reddish calcareous and gypsum clays
PALEOCENE UPPER CRETACEOUS	Mariano Boedo/Yeruá	Sandstone, calcareous, claystones
LOWER CRETACEOUS UPPER JURASSIC	Serra Geral Tacuarembó	Basalts Quartzose sandstones
UPPER TRIASSIC	Buena Vista	Quartzose sandstones
LOWER PERMIAN	Chacabuco	Claystones and Tuffites
UPPER CARBONIFEROUS	Charata Sachayoj	Siltstone and Sandstone Claystones with lutytes and sandstones
PRECAMBRIC	Crystalline Basement	Granite-Gabbro and metamorphic rocks

Figure 2. Stratigraphic chart.

GEOLOGY AND STRATIGRAPHY

Geology and stratigraphy were considered to be fundamental aspects because they are the source of the geohydrological systems. They were exhaustively analysed, taking into account numerous existing publications on the subject – Padula and Mingramm (1968), Russo et al. (1979), Aceñolaza and Sayago (1980), Filí (1983), Tujchneider and Filí (1988), among others. More than 100 well logs were also considered in order to elaborate different geologic cross-sections. A synthesis of all this information is shown in the stratigraphic chart in Figure 2.

GEOMORPHOLOGY

Two large geomorphological units of hydrogeological interest were distinguished. They are called the Paraná Alluvial Valley and Santafesina Remainder Plain. In both of them sub-units were identified.

STRUCTURAL GEOLOGY

On the basis of a review of the existing records (Adina-Spartan, 1977) it was concluded that the studied area had a complex structural conformation and it was affected by the neotectonics. However, it is important to highlight that the numerous structural controls described by the different specialists in their research were not expressed in the well logs analysed or in the geologic cross-sections done, at regional or local level.

SURFACE WATER HYDROLOGY

The drainage pattern of the River Paraná valley section has a north–south direction. The slope of this area and the regional one are not coincident. This suggests a structural control.

For the River Paraná's rates greater than 20,000 m^3/seg, an inflow takes place from its main watercourse towards the west. Rates close to 60,000 m^3/seg determine that the drainage basin of Los Saladillos becomes completely flooded. Consequently, it was not considered to be appropriate to establish a basin boundary in terms of the traditional hydrology procedures, because it was not representative.

In this particular case, the boundary of the drainage area was mobile. In the studied area, the boundary varied in terms of:

- the magnitude of the hydrological processes in the upper Del Plata Basin;
- the possibility that these processes were increased by successive contributions along the whole basin beyond its medium reach;
- the length of local hydrological phenomena in the valley.

The flow regime analysis clearly indicates that Los Saladillos has polimodal and multicomponent behaviour (Figures 3 and 4).

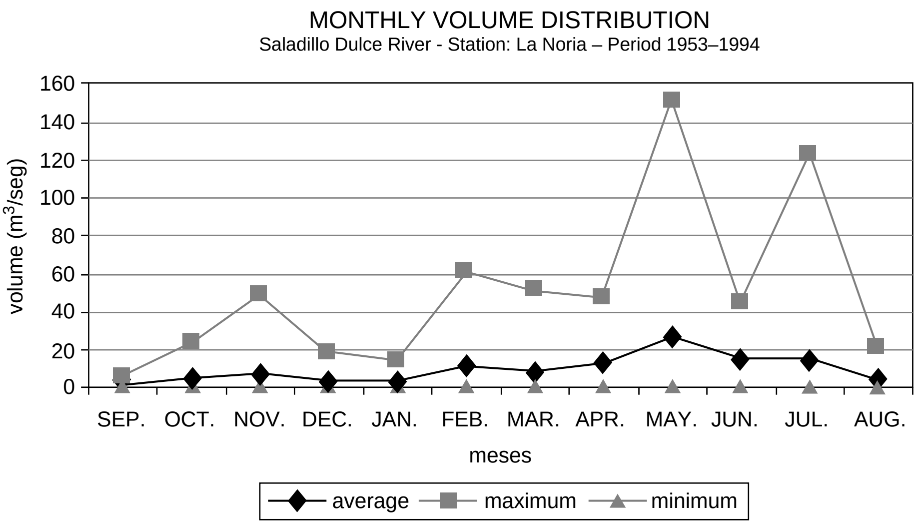

Figure 3. Flow regime Saladillo Dulce River.

HYDROMETEOROLOGY AND WATER BALANCE

The hydrometeorological variables were statistically analysed. Then using the records of Santa Fe's meteorological station the water balance was calculated (Figure 5). This station was selected because it possesses historical data over the longest period (1920–95). These records include the hyper-wet period, which started in the 1970s and

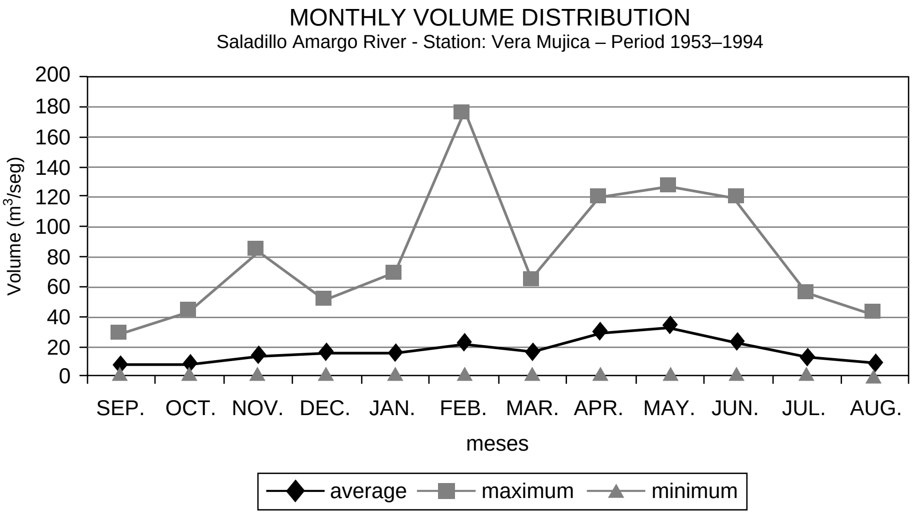

Figure 4. Flow regime Saladillo Amargo River.

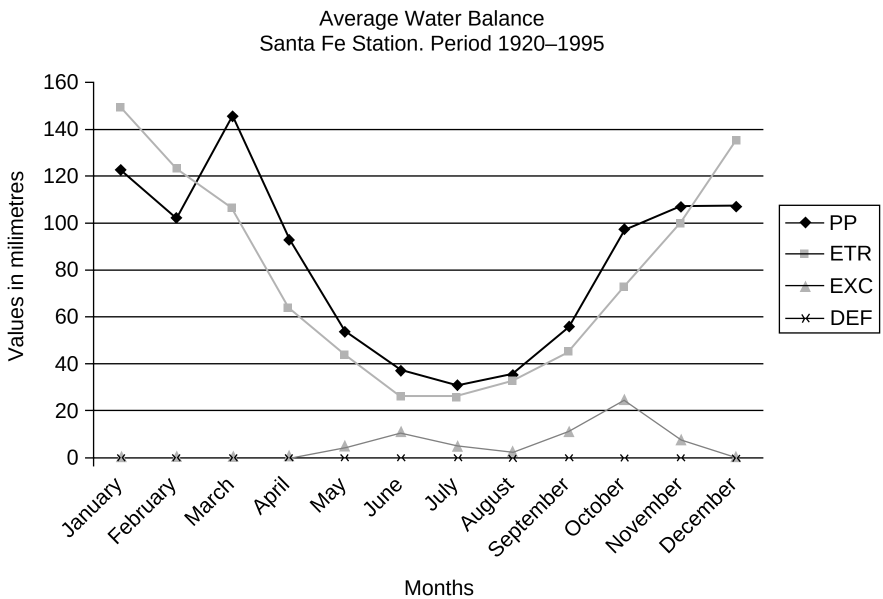

Figure 5. Water balance.

very much affected the area. The average excess of 66 mm/year indicates the water quantity available for runoff and infiltration. The employment of the Thornthwaite-Mather method to estimate evapotranspiration is endorsed by the experiments carried out by González et al. (1997). This demonstrated that it is one of the most appropriate methods in the humid/sub-humid Pampeana Plain. They validated the method of fitting water balances by means of field measurements.

HYDROGEOLOGY

A potentiometric surface map was drawn to plot and contour 256 water level measurements (Figure 6). This map shows a radial surface with flow directions diverging from the recharge areas and converging at the discharge area.

As a result of the morphological analysis of the potentiometric surface map two main recharge areas could be identified as the most obvious features. One is west and the other east of the remainder of the Santafesina Plain. Between these areas, a main central discharge area could also be distinguished, which coincides with the course of Los Saladillos and the lagoon system.

In the western main recharge area, the potentiometric levels decrease from more than 40 m to 25 m. In this area the hydraulic gradient is 1.5 m/km. Between potentiometric levels of about 25 m and 20 m, the average hydraulic gradient decreases to 0.8 m/km. From the 20 m equipotential line up to the discharge line, the hydraulic gradient becomes 0.4 m/km. This fact reflects the groundwater flow through sediments that increase their hydraulic conductivity toward the centre of the valley.

In the main eastern recharge area, the direction of the groundwater flow is from northeast to southwest, with much lower hydraulic gradients (0.2 m/km), indicating the high hydraulic conductivity of the paleo-channels.

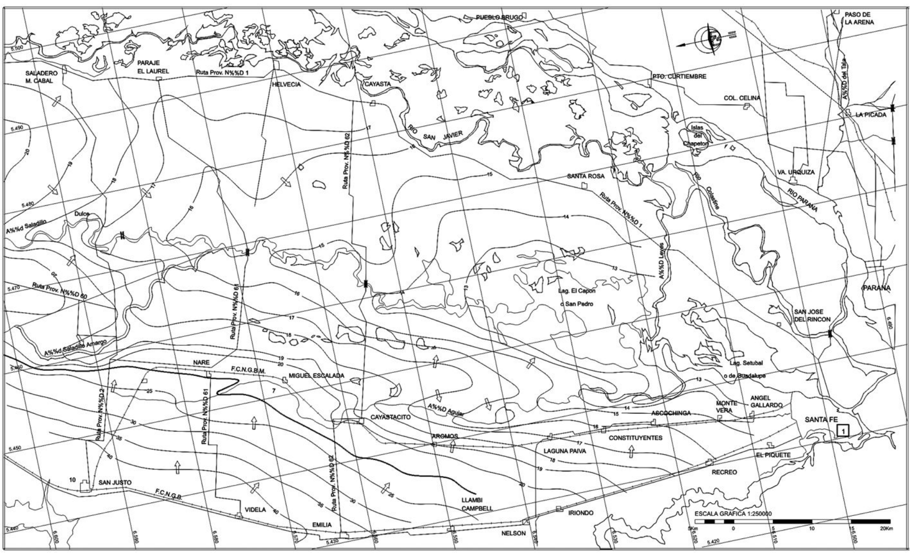

Figure 6. Potentiometric surface map.

HYDROSTRATIGRAPHY

The geological formations in the studied area present regional variations in their constitution that affect the groundwater flow. Consequently, on the basis of well-known stratigraphical units, great hydrostratigraphical units and sub-units were identified. These last units and sub-units govern the groundwater presence, circulation and chemical constitution, as well as its possible link with the external arch of the hydrological cycle.

In the well-known sedimentary sequence, up to approximately 150 m in depth, three main hydrostratigraphical units could be defined and have been named: A, B and C. At the same time, these hydrostratigraphical units show subdivisions of lower hierarchy.

(A) Hydrostratigraphical unit: This is represented by brown and rosy clays, with aggregates of gypsum and carbonates. Grey, green and calcareous clays are superimposed on these brown clays. Due to their clayey constitution and the degree of compaction, the unit hydraulic behaviour is aquiclude. Because of these clays' stratigraphical sequence location, their regional continuity and their thickness, the unit acts as the 'hydrogeological basement'. According to the regional characteristics below this unit, there are no other aquifer units linked directly to the meteoric arch of the hydrological cycle.

(B) Hydrogeological unit: This conforms to a very complex multi-layer aquifer system. This unit consists of sands with clays intercalations corresponding to the Paraná Formation and to the River Paraná's alluvial valley deposits.

(C) Hydrogeological unit: Different deposits of the clayey-silty fraction constitute this unit. It is predominantly aquitard. This unit constitutes the upper portion of the sedimentary column.

AQUIFER HYDRAULICS

Both field pumping tests and laboratory analysis were carried out in order to obtain and adjust the formational hydraulic parameters. Theis and Cooper-Jacob methods were used in confined aquifers. Neuman's method was used in unconfined aquifers with later drawdown effects. For this aquifer system a K average value of 50 m/day was considered representative.

Figure 7 shows a synthesis of the hydraulic tests results. The map on Figure 8 indicates the hydraulic conductivity spatial variability. The highest value (70 m/day) was measured

Site	*Well Nº*	*T (m^2/day)*	*K (m/day)*	*S*
Site Nº 2	1	786	20	$3{,}50\times10^{-3}$
	2	1230	30,7	$1{,}20\times10^{-3}$
Site Nº 3	1	1031	34,0	$1{,}74\times10^{-2}$
	3	1350	44,9	$8{,}33\times10^{-3}$
Site Nº 4	2	1470	49,0	$2{,}01\times10^{-3}$
	4	1312	43,6	$1{,}83\times10^{-3}$

Figure 7. Hydraulic tests results.

at the sandy saddle pad at the east of the valley. The K value decreases to a minimum of 20 m/day for clayed silty sediments at the central-western sector of the area.

HYDROCHEMISTRY

A database was constructed using the available information defined by a matrix of 106 observations with 11 variables describing the main physicochemical characteristics of the samples. These characteristics were pH, electric conductivity, TDS, Na^+, K^+, Ca^{++}, Mg^{++}, $CO_3^=$, CO_3H^-, Cl^- and $SO_4^=$.

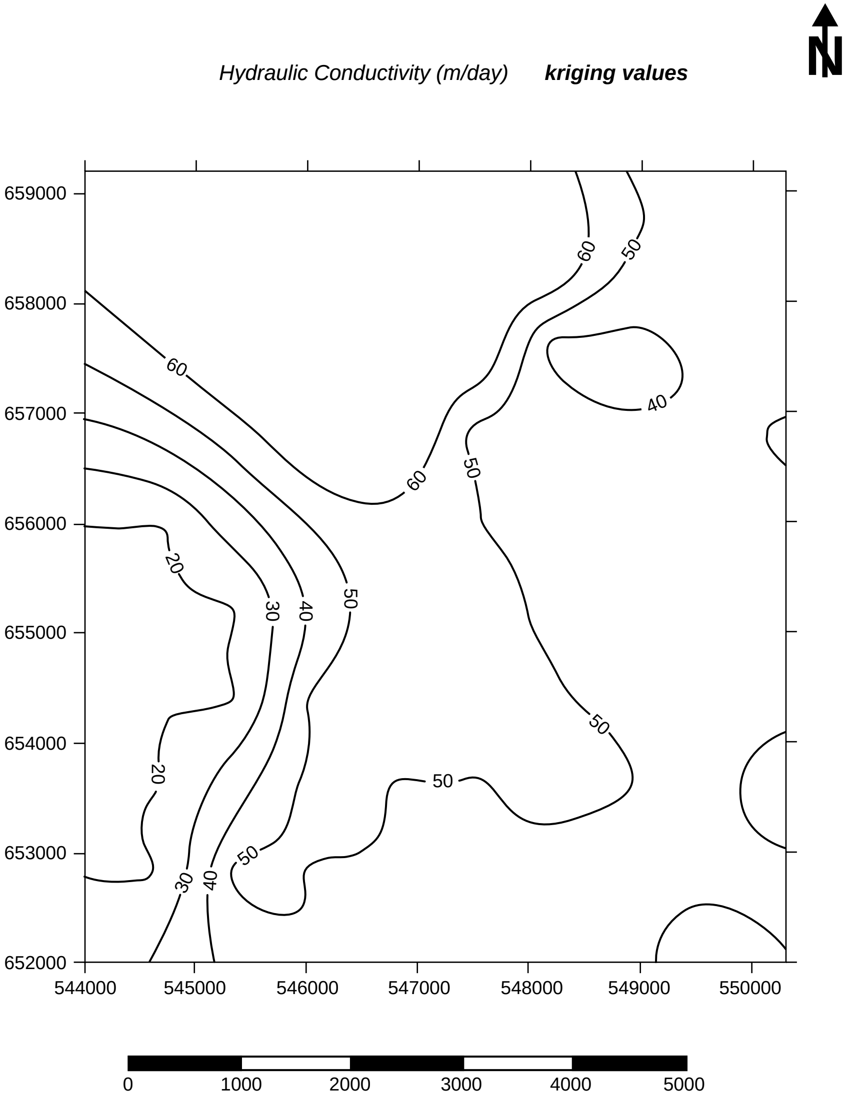

Figure 8. Hydraulic conductivity spatial variability.

In these particular environments, it was observed how the groundwater chemistry was an excellent hydrodynamic indicator. At the same time, it had to be linked to the hydrological characters and those which could be noticed from the physical environment (geomorphology and hydrostratigraphy). Figures 9 and 10 show the spatial variation of Cl^- and $SO_4^=$ in the studied area, respectively.

Figure 9 shows a clear increment of Cl^- content that follows the flow direction. It reaches its maximum at the discharge area. Figure 10 shows similar behaviour for SO_4.

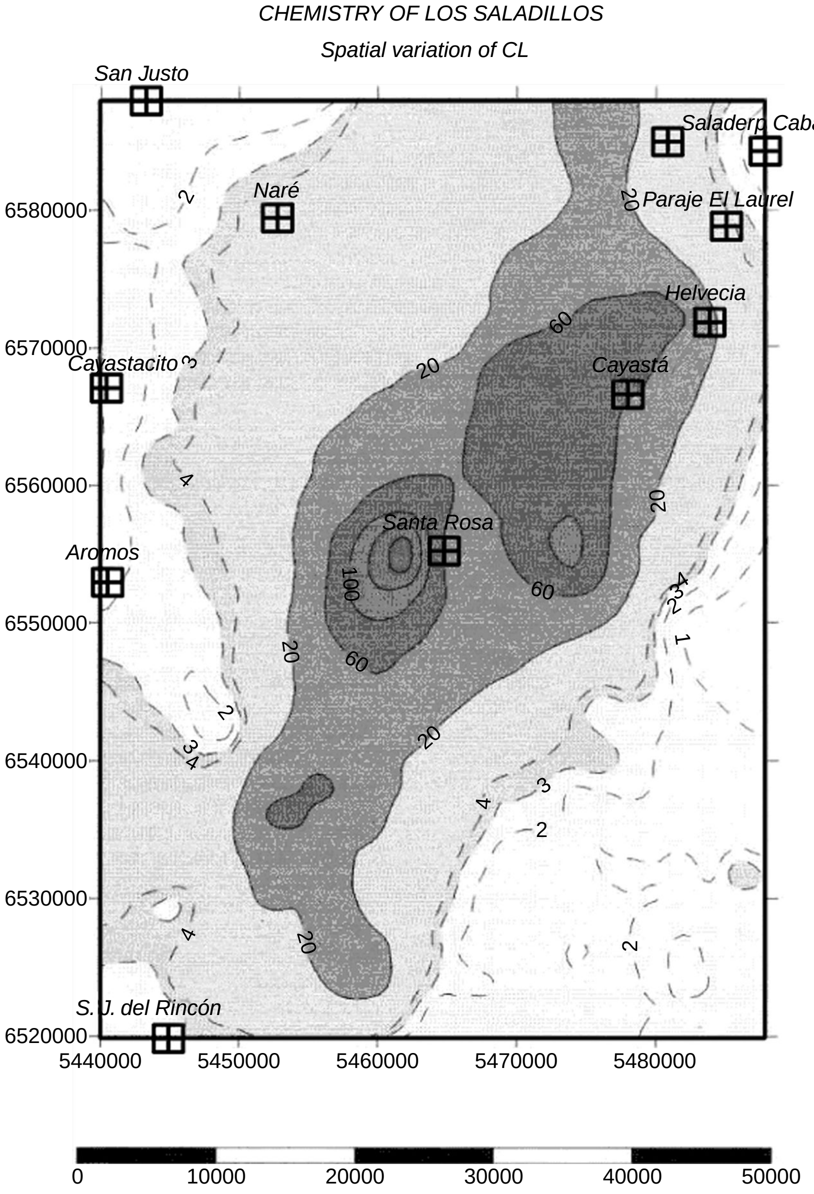

Figure 9. Spatial variation of Cl^-.

On the other hand, CO_3H^- presents its highest values in the main recharge areas and it diminishes relatively towards the discharge area.

Cations accompany this evolution, showing that the alkalis have similar behaviour to the chlorides and the alkaline earths in relation to the bicarbonates.

The application of R-mode cluster analysis to this data set, leads to the conformation of two main groups: i) Cl^-, $Na^+ + K^+$ as the main core to which $SO_4^=$ is associated and ii) pH, CO_3H^-. In factorial analysis by principal components, two factors are

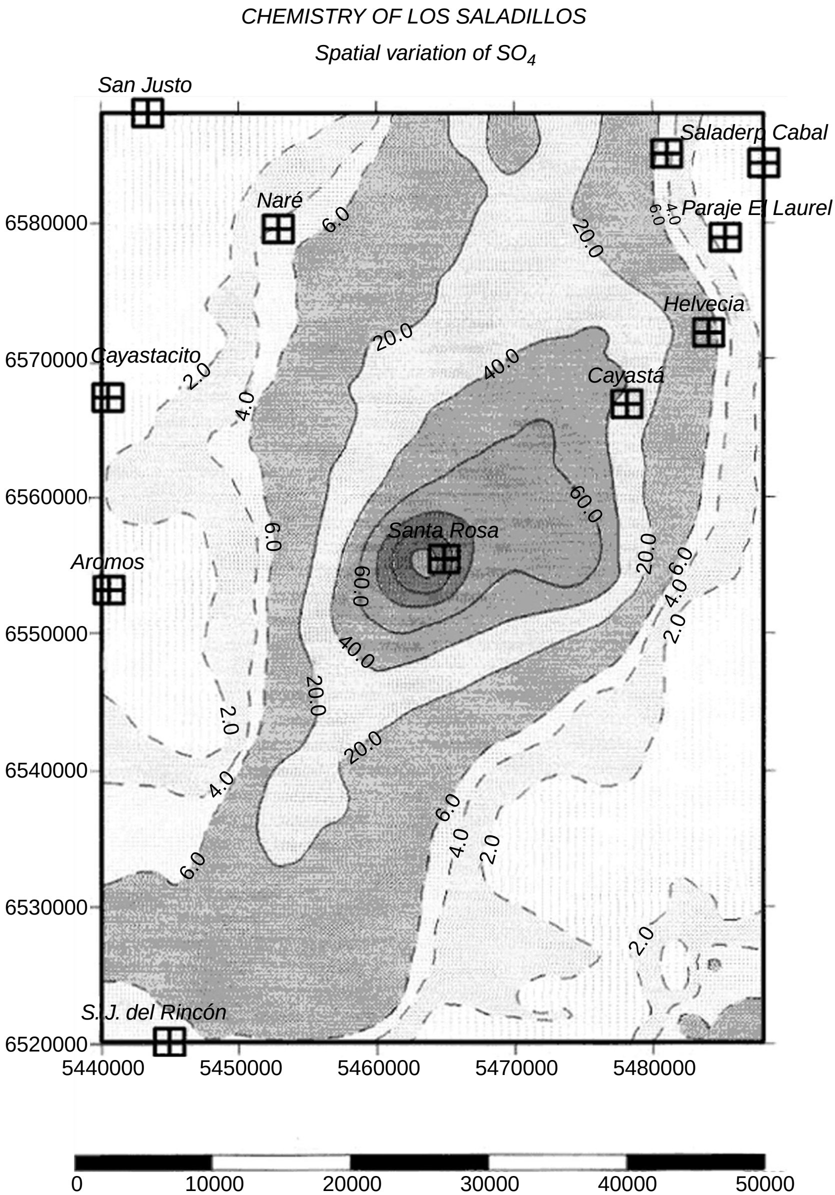

Figure 10. Spatial variation of $SO_4^=$.

retained: i) Factor 1 is defined by Cl^-, TDS, $SO_4^=$, $Na^+ + K^+$, Mg^{++} and Ca^{++}; ii) Factor 2 is defined by pH and CO_3H^-. Both factors represent 90 per cent of the total variance.

The hydrochemistry behaviour supported the results that came from the hydrological and hydrodynamic analysis. This fact is especially evident when corroborating the spatial occurrence of the recharge and discharge areas, the groundwater flow direction and the mostly effluent character of the streams.

It is in relation to these last points that a further judgement element, given by the hydrochemistry and mentioned in the local toponymy, was added; the denomination of Saladillo Amargo Creek (Bitter Saladillo) makes mention of the sulphated character of its water. In Figure 10 it can be seen that the increment of the ion $SO_4^=$ takes place in the direction of this streambed, ratifying its effluent behaviour.

The toponym Saladillo Dulce Creek (Sweet Saladillo) shows the presence of bicarbonates and low levels of chlorides in solution. The geomorphological features are coincident with the whole situation. The Saladillo Dulce Creek flows through psamitic mounds of large packed saddle sands. The Saladillo Amargo Creek flows on an elongated sector of depressions, facilitating the processes of the groundwater flow discharge. The evaporation process also takes place in this area when the phreatic level outcrops or is near the ground surface.

CONCEPTUAL MODEL DEFINED

The convergence of thematic evidence allows the definition of an aquifer system integrated by two subsystems: i) an aquifer of shallow groundwater and ii) an aquifer of deep groundwater, both of them interrelated in the Los Saladillos Basin.

The elaborated conceptual model is presented in Figures 11 and 12.

The geomorphology and lithological constitution of the upper geological formations control the hydrodynamic and hydrochemistry of the shallowest subsystem. In this subsystem, groundwater recharge, conduction and discharge areas have been recognized. They are identified as follows.

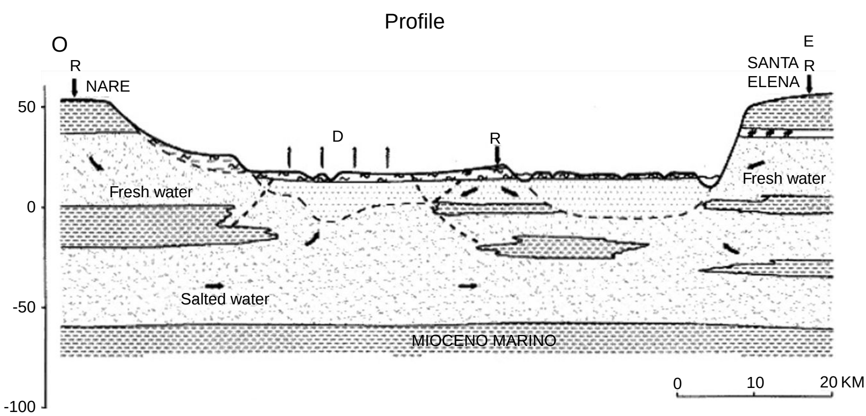

Figure 11. Conceptual model.

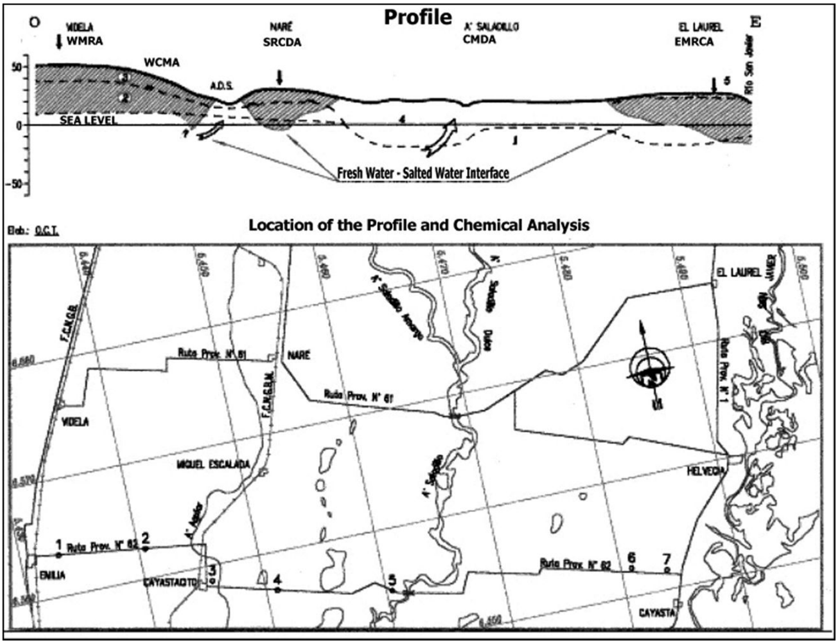

Figure 12. Conceptual model.

Western main recharge area (WMRA): This is composed of a strip of land with a northeast–southwest direction, 5–10 km wide and with a 50–60 m elevation. It coincides with the non-dissected Santafesina Reminder Plain surface and acts as the divide between the shallow groundwater and the surface water.

Rainfall is concentrated on the circular depressions that characterize the geomorphology of the area. When it reaches saturation level it constitutes the groundwater recharge. The upper part of the stratigraphic column is composed by approximately 20 m of silts and clayey silts that are hydraulically aquitards.

Western main conduction area (WMCA): This belongs to the colluvial dissected slope of the Santafesina Remainder Plain and appears as a strip of land with an elevation of 40–50 m. The thickness of the clayey-silty cover is approximately 8 m and hydraulic conductivity is low.

Central main discharge area (CMDA): Below an elevation of 22 m there is a strip of land 30 km wide that has its geomorphological correspondence with the central plain of the Paraná River Valley. This plain consists of flood plain depressions, obliterated accumulation forms, integrated paleodrainage, permanent courses and lagoons.

Los Saladillos Creek is the main permanent watercourse. Modern alluvial silts of varied hydraulic conductivity cover the basin. The general direction of the groundwater flow is toward the axis of the area along which both Saladillos flow. The deep subsystem also partially discharges in this area.

Eastern main recharge and conduction area (EMRCA): This is composed of a strip of land of irregular width, which is bounded to the east by the MCDA and acts as the surface divide between Los Saladillos and the River Paranás current valley.

Hydrogeological Units	TDS [mg/l]	CO_3H	SO_4	Cl	Ca	Mg	Na	K
Western main recharge area	170	3.54	0.19	0.45	1.56	0.84	1.60	0.18
Secondary recharge conduction and discharge area	1.350	10.62	1.60	6.23	8.11	2.49	7.60	0.31
secondary recharge	700	7.43	0.82	0.45	2.70	0.80	5.00	0.20
Conduction and discharge area	4.300	7.91	17.50	40.30	9.40	1.60	54.30	1.20
Central maina discharge area	26.200	9.73	88.50	312.00	36.40	51.10	320.00	3.40
Eastern main recharge and conduction area	360	4.43	0.56	8.82	2.97	0.53	10.00	0.25
Eastern main recharge area	200	3.89	0.21	0.61	2.12	0.68	0.40	0.17

Figure 13. Representative hydrochemical characteristics of the different environments defined.

Here, the groundwater divide moves to the west when the river floods. The area is a portion of the alluvial plain. It consists of very fine sands with a thickness of almost 5 m. These sands lie on clayey-silty sediments that constitute an independent and discontinuous aquifer level.

Secondary recharge, conduction and discharge areas (SRCDA): These are associated with the geomorphological terrace units. The water integrating the shallow groundwater subsystem flows and discharges in flood-plain depressions, mainly contributed to by the local streams. They constitute a base level for the subsurface water infiltrated in the WMRA and are not able to reach the deepest levels.

Los Saladillos terrace is located between both Saladillos, as a narrow strip. It behaves as a recharge and conduction area. The discharge is towards both courses. The geological material of these areas consist of fine slimey-loamy and sandy silts with aquitard behaviour.

The deep system is not easily detectable in terms of the relationship of the hydraulic heads. Only if the wells are appropriately drilled and the upper system is correctly isolated it is possible to quantify the differences between the potentiometric levels of the lower and the upper systems, since the former overcomes the latter in very small values. This is due to the characteristics of the slope of the flatlands. Sometimes abrupt changes in the water saline content are considered to be a horizontal zonation, corresponding to the same level, when in fact they are due to two differentiable subsystems, which are interrelated in Los Saladillos area.

Figure 12 shows an east–west hydrogeological cross-section in which the stratigraphical conformation has been outlined as a fresh-salty water interface and conceptual hydrodynamic operation. The cross-section references are: 1 – Paraná Formation, 2 – Ituzaingó Formation, 3 – Pampa group, 4 – Alluviums and 5 – Dunes.

Figure 13 shows the representative hydrochemical characteristics of the different environments defined. The most probable anions and cations contents are expressed in meq/L.

The conceptual model arises from a critical path: the hypotheses presented are supported by the real convergence of thematic evidence. So, when several pieces of evidence point to the same conclusion, we have the most commendable hypothesis. That is to say, the one that offers the smallest uncertainty threshold.

This reasoning gives solidity to the conceptual model, keeping in mind that the hydrological events occurred in a scenario of very difficult resolutory boarding – sub-humid or humid plains.

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CHAPTER 13

Reference evapotranspiration in the River Azul Basin, Argentina

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ABSTRACT: Remote sensing is among the latest techniques to assess the spatial and time-dependent variations of evapotranspiration (ET). The newest models based on energy balance approaches are able to come up with precise estimates of ET at any point of the time-space domain. However, those models have to be fed by information that is not always available. This paper presents a simpler approach that allows the estimation of ET by combining satellite information and data from a weather station. The model needs to rely on images of surface temperature (T_s) and to estimate ET from regular meteorological data and methods. Its application to the River Azul Basin – 6000 km^2 in the centre of Buenos Aires Province, Argentina – led to a precision of ± 0.4 mm/day (11 W/m^2) with a correlation coefficient (r^2) of 0.88. The model was applied to an image of T_s and the results compared with what was obtained from data of a reference weather station. The model gave an estimate of 4.2 mm/day (119 W/m^2), whereas the conventional method rendered a value of 4.5 mm/day (128 W/m^2). It is concluded that the proposed approach is able to deliver reliable data on the spatial and temporal variations of ET in areas where the availability of data is scarce.

INTRODUCTION

Input data for hydrological regional models consist of a large number of variables for several time periods. In order to describe properly the features of large basins, such hydrological models should be distributed, that is, able to take into account the spatial distribution of variables (Sandholt et al., 2002).

Evapotranspiration (ET) is one of the most important output variables from hydrological models as far as its distribution in time and space is concerned. Classic methodologies for ET estimation do not handle properly such time-space variations. ET

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estimates are obtained at discrete sites and their regionalization is made on the basis of interpolation, which is not representative of what actually occurs.

More recently, models for ET estimation that solve the energy balance equation from remote sensing data have been developed (Boegh et al., 2002). They make it possible to obtain the space and time variations of ET by carrying out energy balances at soil level (Schmugge et al., 2001). The remote sensing technique becomes a useful tool for retrieving information on the energy and water balance, and is able to detect changes at the regional or even the parcel levels. The unit area for which the balance is carried out depends on the spatial resolution of the sensor used. For instance, for the Advance Very High Resolution Radiometer (AVHRR) and Moderate Resolution Imaging Spectroradiometer (MODIS), the working unit is one km^2; for the LANDSAT Thematic Mapper (TM) 5 the area is 14,400 m^2 and 3600 m^2 for the LANDSAT-ETM+7. Thus, remote sensing – thoughtfully employed – promotes an adequate knowledge of water losses from the system, which, in turn, can be used to improve the management of regional water resources.

Energy balance-driven models need to count on net radiation data, which are a type of information that are not always readily available. This paper aims at assessing the spatial variation of ET in an area of scarce data. To do so, a linear model for determining the reference ET from surface temperature data (T_s) at the pixel level was applied. The model is based on the split window approach (Coll & Caselles, 1997).

The model requires the availability of enough meteorological information to compute ET with the Penman-Monteith equation, as well as series of T_s images to adjust its parameters.

A detailed description of the steps followed to adjust the model parameters is presented and its application to the River Azul Basin is shown. The water balance in such a basin, located at the centre of Buenos Aires Province, Argentina, indicates that ET is the single largest component inasmuch as it accounts for about 90 per cent of the water that enters the system as annual rainfall (Usunoff et al., 1999).

METHODOLOGY

A linear model that combines remote sensing data with the Penman-Monteith evapotranspiration equation was used to estimate the reference ET (ET_o mm/day) at the pixel level. The model is generated by splitting the Penman-Monteith equation into two terms: the radiation and the aerodynamic components. Rearranging the equation leads to the following linear model:

$$ET_o = \boldsymbol{a} \cdot T_s + \boldsymbol{b} \tag{1}$$

where T_s is the surface temperature (°C), and ***a*** (mm/day/°C) and ***b*** (mm/day) are parameters specific to the study area.

The ***a*** parameter represents the radiation effects on the surface and is quite sensitive to changes in the wind speed. The ***b*** parameter takes into account the effect of meteorological conditions on the surface.

The model assumes a flat surface covered by vegetation 0.12 m high and with enough water availability. Besides, the meteorological variables are supposed to show spatial homogeneity.

The information required by the model is rather minimal: satellite images from which T_s can be retrieved and meteorological data to estimate ET_o by means of the Penman-Monteith equation.

Figure 1 displays a simplified scheme of the steps that have to be followed to obtain the ***a*** and ***b*** parameters.

By combining the information stored in the various bands of the AVHRR-NOAA images, the normalized difference of vegetation index (NDVI) can be determined, as well as the percentage of vegetation cover (P_v) and the emissivity (ε). Knowing the emissivity and the vapour content of the atmosphere and the surface temperature (bands 4 and 5), the estimation of T_s is carried out by using the split window approach.

With daily meteorological information, ET_o can be calculated. The data needed to do so are: maximum and minimum air temperature ($T_{a\ max}$ and $T_{a\ min}$), maximum and minimum air relative humidity (HR_{max} and HR_{min}), number of hours of sunshine (n) and wind speed at 2 m height (U_2).

After calculating T_s and ET_o, it is then possible to adjust the ***a*** and ***b*** parameters. Following that step, and by applying Equation 1 to images of T_s, maps of ET_o can be obtained.

Estimation of ET_o

The estimation of the daily reference evapotranspiration (ET_o) was done by employing the Penman-Monteith-FAO equation (Allen et al., 1998):

$$ET_o = \frac{0.408 \cdot \Delta \cdot (R_n - G) + \gamma \cdot \frac{900}{T_a + 273} U_2 \cdot (e_s - e_a)}{\Delta + \gamma \cdot (1 + 0.34 \cdot U_2)} \tag{2}$$

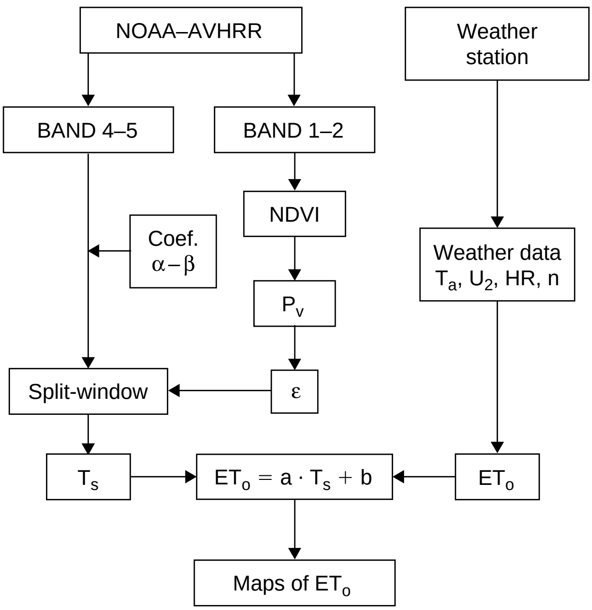

Figure 1. Schematic diagram of the proposed model (symbols are defined in the text).

where ET_o is the reference evapotranspiration (mm/day), R_n is the net radiation (MJ/m^2/day), G is the soil heat flux density (MJ/m^2/day) (disregarded because of the time scale considered), γ is the psychrometric constant (kPa/°C), T_a is the air temperature at 2 m height (°C), U_2 is the wind speed at 2 m height (m/s), e_s is the saturation vapour pressure (kPa), e_a is the actual vapour pressure (kPa), and Δ is slope of the water vapour pressure curve (kPa/°C).

Before calculating ET_o, a consistency analysis of the meteorological variables recorded at the weather station was made. To do this, data from the reference weather station were compared (homogeneity check) with data from three other weather stations in the area. The variables analysed and compared were $T_{a\,max}$, $T_{a\,min}$, U_2, n, HR_{max} and HR_{min} at daily time intervals and for the same time period. The mean, standard deviation, covariance, regression equations, correlation coefficients and residuals were calculated and evaluated. Those values that were deemed not consistent were eliminated from the records.

ET_o was calculated using meteorological data from the reference weather station (Azul) for the period 1996–2000. Mean weekly estimates of ET_o were obtained by processing the satellite images.

Calculation of T_s

T_s was calculated using the split window approach, which accounts for the atmospheric and emissivity correction based on the difference in atmospheric absorption recorded in the radiances of bands 4 and 5 of the AVHRR sensor (Coll & Caselles, 1997). Radiation measured in such bands were converted into brightness temperature (noted as T_4 and T_5) by using the calibration equation provided for the sensor.

Estimates of T_s used the quadratic split window equation proposed by Coll and Caselles (1997):

$$T_s = T_4 + [1.34 + 0.39 \cdot (T_4 - T_5)] \cdot (T_4 - T_5) + 0.56 + \alpha \cdot (1 - \varepsilon) - \beta \cdot \Delta\varepsilon \qquad (3)$$

where T_s is the surface temperature (K) in the pixel, T_4 and T_5 are the brightness temperatures from bands 4 and 5 of NOAA-AVHRR (K), α and β are coefficients that depend on the water vapour content in the atmosphere (K), ε is the surface emissivity in the range 10.5–12.5 μm, and $\Delta\varepsilon$ is the difference in emissivity between bands 4 and 5.

The algorithm is simple in that it requires knowledge of the water vapour content in the atmosphere to determine the α and β coefficients. The standard error for T_s is $\pm 1.0 - 1.5$ K for atmospheres with low to moderate humidity.

The α and β parameters in Equation 3 correspond to atmospheres of medium latitude in winter and summer. Inasmuch as α does not vary with the water vapour content in the atmosphere, a value of 50 K was employed. The β coefficient decreases linearly with the water vapour content of the atmosphere and values of $\beta = 75$ K for summer and $\beta = 150$ K for winter were adopted.

ε was assessed from the equation proposed by Valor and Caselles (1996) for the spectral range of 10.5–12.5 μm:

$$\varepsilon = \varepsilon_v \cdot P_v + \varepsilon_s \cdot (1 - P_v) + 4 \cdot d\varepsilon \cdot P_v \cdot (1 - P_v) \qquad (4)$$

where ε is the effective emissivity of a surface made up of soil and vegetation, ε_v is the vegetation emissivity, P_v is the percentage of vegetation cover, ε_s is the soil emissivity and dε is the cavity term that takes into account the internal reflections between the vegetation and the soil.

The percentage vegetation cover (P_v) was calculated from the NDVI using the equation obtained from a linear model of reflectivity with two components (soil and vegetation) and two bands (red and near infrared), according to Valor and Caselles (1996):

$$P_v = \frac{(1 - \frac{NDVI}{NDVI_s})}{(1 - \frac{NDVI}{NDVI_s}) - K \cdot (1 - \frac{NDVI}{NDVI_v})} \tag{5}$$

being $K = \dfrac{IRC_v - R_v}{IRC_s - R_s}$

where IRC_v is the vegetation reflectivity in the near infrared, IRC_s is the soil reflectivity in the near infrared, R_v is the vegetation reflectivity in the red, R_s is the soil reflectivity in the red, $NDVI_v$ corresponds to pure vegetation, $NDVI_s$ corresponds to pure soil and NDVI is the vegetation index of any considered mixed pixel.

For the study area, Equation 4 reduces to the first two terms because no internal reflections are expected due to the type of vegetation cover. Indeed, the vegetation cover consists mainly of natural pastures and agricultural crops (wheat, corn, soybean) whose cavity effects are minimal.

For the determination of ε the following values were used in Equation 4: 0.985 for vegetation emissivity and 0.96 for bare soil emissivity. The ε_v value matches field data from Rubio et al. (1997), whereas the ε_s value corresponds to data reported by Salisbury and D'Aria (1992, 1994).

T_s estimates used 21 images captured by the AVHRR sensor during the period 1992–2000, of 1.1 km^2 pixel size for different seasons throughout the year. The selection of weekly images took into account the following constraints: that the maximum daily air temperature at the reference weather station had to differ at least 2°C from the mean of the weekly maximum temperatures, it had to be a cloudless day ($n \cong N$) and there had to be no rainfall in the previous days. Images of dry and humid periods were used in order to cover the maximum and minimum values for T_s. Each image selected and processed represented the mean weekly T_s.

Using weekly images increases the probability of making use of high-quality radiometric data, which is essential for the T_s estimation.

The T_s value was taken as the average of an area of 3 by 3 pixels, where the central pixel corresponded to the location of the reference station. Such an average comes from the geometric correction of the image, whose error is ±1 pixel.

Determination of parameters *a* and *b*

To determine ***a*** and ***b***, it is imperative to know T_s and ET_o for the area where the reference weather station is located. The expression for ***a*** and ***b*** comes from the following equations, which assume that the Penman-Monteith equation can be written as the sum of two terms: aerodynamic (ET_{aero}) and radiation (ET_{radio}):

$$ET_o = ET_{aero} + ET_{radio} = \boldsymbol{a} \cdot T_s + \boldsymbol{b} \tag{6}$$

By rearranging Equation 6 it is possible to obtain explicit equations for ***a*** and ***b***:

$$a = 0.408 \cdot \omega_{radio} \cdot \varepsilon_s \cdot \sigma \cdot A \tag{7}$$

$$b = ET_{aero} + 0.77 \cdot R_s + 0.408 \cdot [\varepsilon_s \cdot \sigma \cdot (B - \varepsilon_a \cdot (T_a)^4)] \cdot \omega_{radio} \tag{8}$$

where ω_{radio} (defined below) is in mm m^2/MJ, ε_s is the surface emissivity, σ is the Stefan Boltzmann constant (4.9×10^{-9} MJ/m^2/K^4/day), A is equal to 1.1×10^8 K^3, 0.408 is the conversion factor to mm/day, ET_{aero} is the aerodynamic term of Penman-Monteith equation (mm/day), R_S is the solar radiation (MJ/m^2/day), 0.77 $(1 - \alpha)$ is a value that comes from assuming a surface albedo α of 0.23, B equals -2.46×10^{10} K^4, ε_a is the air emissivity and T_a is the air temperature (K). ω_{radio} has the following form:

$$\omega_{radio} = \frac{\Delta}{\Delta + \gamma \cdot (1 + 0.34 \cdot U_2)} \tag{9}$$

where Δ is the slope vapour pressure curve, (kPa/°C), γ is the psychrometric constant (kPa/°C), 0.34 is a conversion factor for γ and U_2 is the wind speed at 2 m height (m/s).

The ***a*** parameter represents the radiation effects on the surface and is quite sensitive to changes in the wind speed. For a given radiation, as the wind increases, ***a*** decreases linearly. The ***b*** parameter represents the effects of the meteorological conditions, being sensitive to air temperature and relative humidity, to wind speed and to sunshine. In general, ***b*** decreases linearly as the air temperature and relative humidity increase. Likewise, ***b*** shows a logarithmic behaviour as the wind speed increases and increases linearly as sunshine increases. The sunshine effect is minimum since counting on good images implies cloudless days.

Figure 2 depicts a simplified scheme used to obtain the parameters at the reference weather station.

Values for the ***a*** and ***b*** parameters were obtained from the T_s–ET_o data at the reference weather station, given all the images available and a linear fit using least squares.

Having fitted the ***a*** and ***b*** parameters, Equation 1 was applied to a T_s image corresponding to the second week of January 1996. Finally, the ET_o calculated with data from the weather station was compared to that given by the model (average of a 3 by 3 pixels). The same procedure was applied for each T_s image used.

Study area

The study area corresponded to the River Azul Basin, located at the centre of Buenos Aires Province, Argentina (Figure 3). It covers about 6000 km^2 of Argentina's humid pampas, essentially a flat landscape. The mean regional slope is less than 1 per cent, even reaching values as low as 0.2 per cent at the northern portion (Varni & Usunoff, 1999). The climate is humid, with low to zero water deficit, a mean annual precipitation of 950 mm and a mean annual ET of 915 mm (Usunoff et al., 1999; Varni et al., 1999).

RESULTS

Figure 4 displays the resulting P_v by applying the linear model of reflectivity with two components from the image of the second week of January 1996. It can be observed that the basin shows a high percentage of vegetated areas.

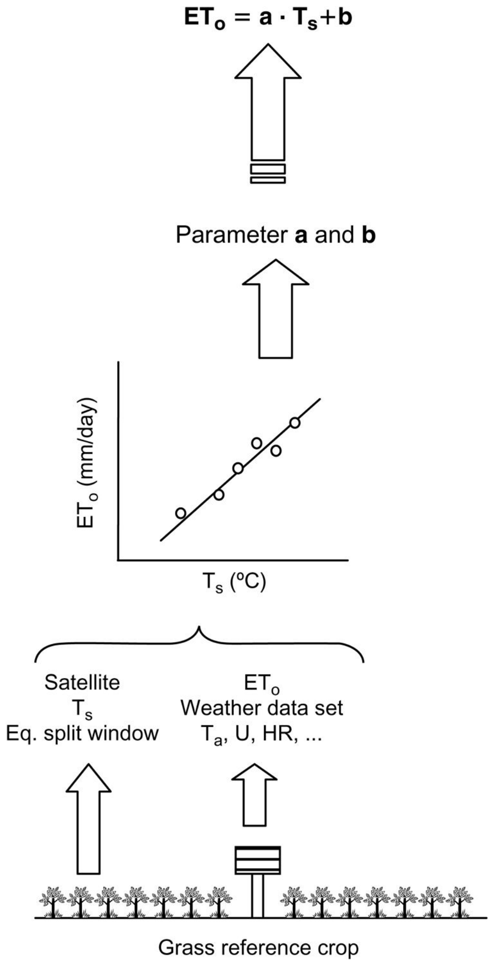

Figure 2. Simplified diagram of fitted parameters.

Figures 5a, 5b and 5c show the distribution of ε_4, ε_5 and ε obtained with Equation 4, and the ε_v and ε_s corresponding to vegetation and soil.

The surface temperature, T_s (°C), obtained with the quadratic approximation split window, is shown in Figure 6.

Figure 7 is a plot of the daily ET_o values that were calculated for the period 1996–2000 at the reference weather station in Azul.

The T_s values in Figure 8 were calculated by applying Equation 3 and the average weekly ET_o was obtained from the daily values. The linear function shown fits the T_s and the ET_o values.

The extreme values in Figure 8 correspond to seasons of maximum and minimum ET_o (summer and winter), whereas intermediate values correspond to autumn and spring seasons.

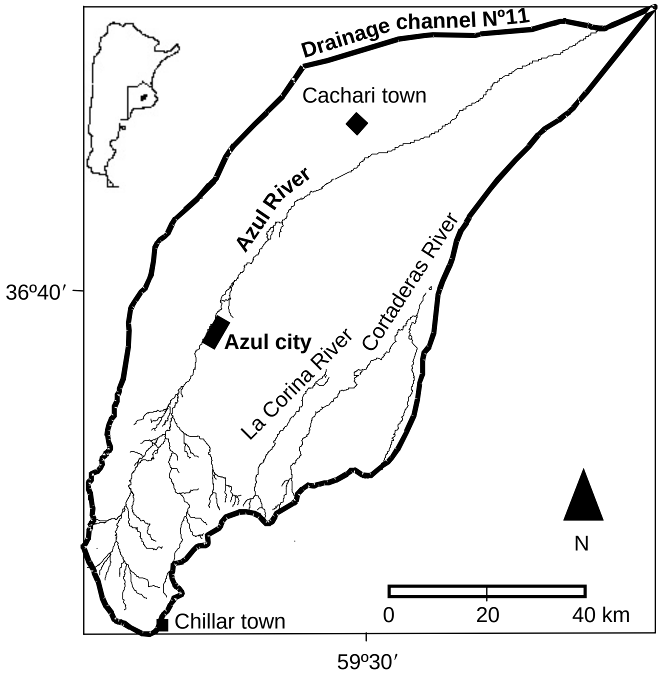

Figure 3. Location of the study area.

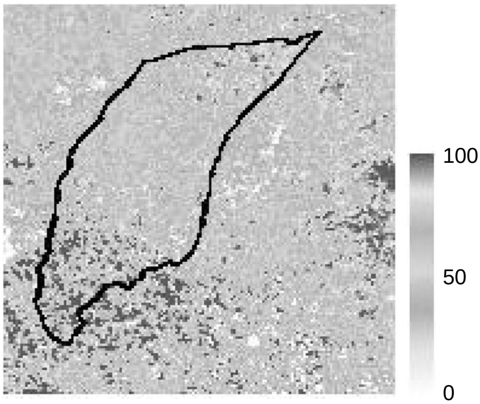

Figure 4. Vegetation cover map (%).

The following equation contains the ***a*** and ***b*** parameters coming from the linear fitting of the points in Figure 8 (using least-squares):

$$ET_o = 0.12 \cdot T_s - 0.3 \tag{10}$$

The fit is quite acceptable ($r^2 = 0.88$), with an estimation error of ± 0.4 mm/day ($11\ W/m^2$).

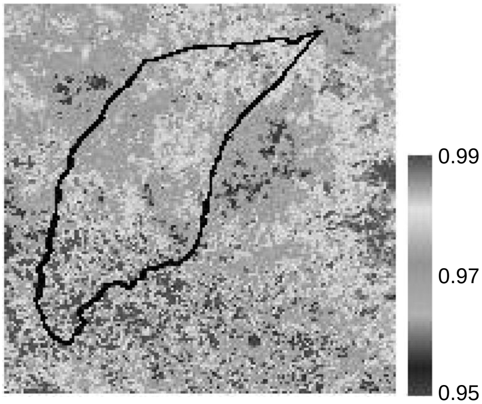

Figure 5a. Emissivity map in band 4.

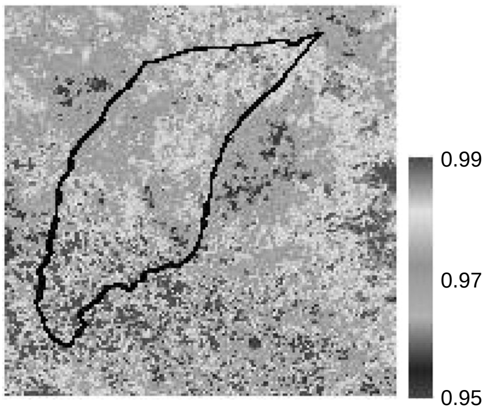

Figure 5b. Emissivity map in band 5.

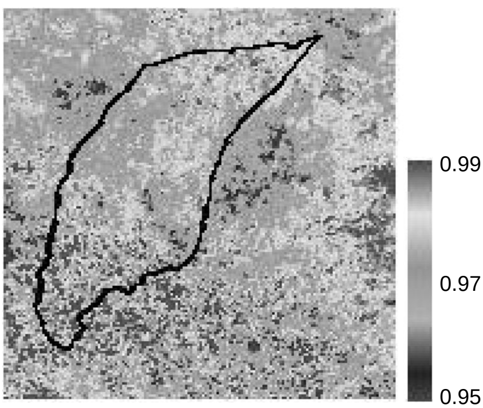

Figure 5c. Emissivity map in branch spectral 10.5−12.5 μm region.

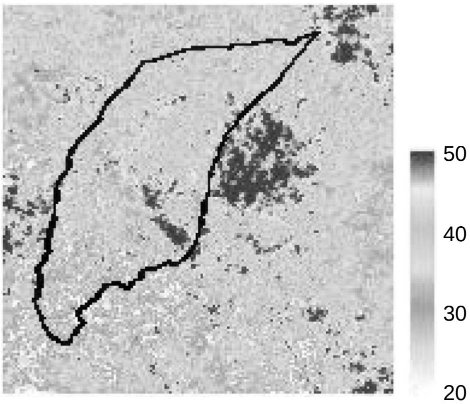

Figure 6. Surface temperature in the Azul Basin (°C).

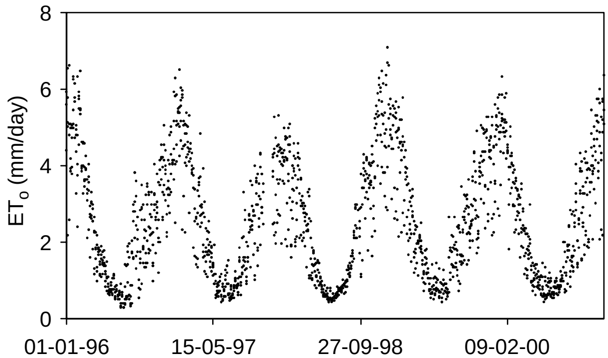

Figure 7. Daily ET_o for the period 1996–2000 in the reference weather station.

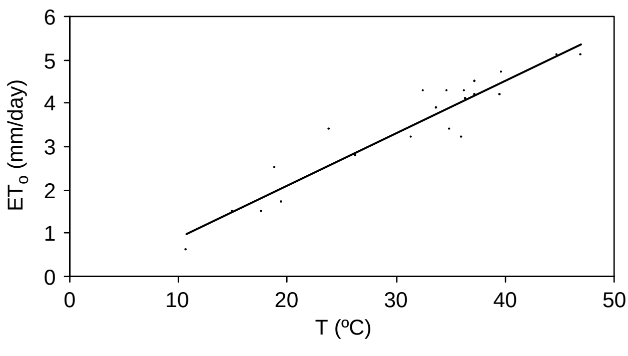

Figure 8. Linear fit of T_s–ET_o data.

Results of the proposed model (Equation 10), as applied to the T_s image of the River Azul Basin are shown in Figure 9. The mean weekly ET_o for the reference weather station was 4.2 mm/day (119 W/m^2), whereas the model gave an estimate of 4.5 mm/day (128 W/m^2).

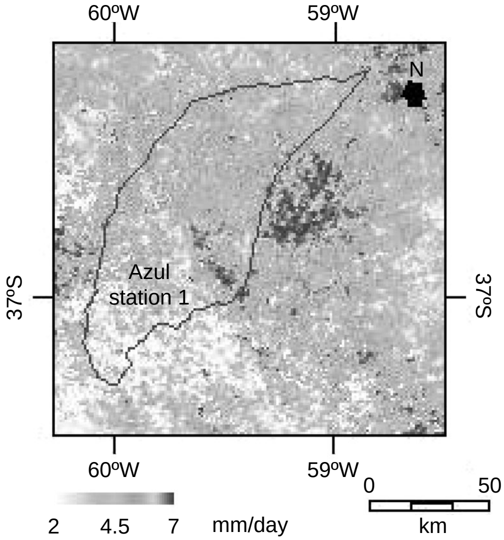

Figure 9. Mean daily ET_o (mm/day) obtained from the model.

CONCLUSIONS

The model presented allows the estimation of weekly ET_o from the T_s information retrievable from a AVHRR-NOAA sensor. Images of T_s should be available, as well as meteorological data to calculate ET_o.

Nowadays, series of satellite images to calculate T_s are readily available and the meteorological information required by the model is rather modest, so its application can be easily implemented.

The procedure to determine the model's parameters is simple and from them it is relatively easy to calculate ET_o at the regional level.

The methodology proposed here leads to the assessment of the spatial and time-related variations of ET_o, which can be picked up later on as input for distributed hydrological models. As a matter of fact, that was the main objective of this project.

As an example, the model has been applied to the River Azul Basin, Buenos Aires Province, Argentina. The parameters were fitted by a function with an r^2 of 0.88 and an estimation error of ± 0.4 mm/day (11 W/m^2).

By using a T_s image of the study area, the mean weekly ET_o turned out to be 4.5 mm/day (128 W/m^2), whereas the corresponding value at the reference weather station was 4.2 mm/day (119 W/m^2), values that are close enough to indicate a good performance of the model.

ACKNOWLEDGEMENTS

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CHAPTER 14

Buried valley ribbon aquifers: A significant groundwater resource of south west Ireland

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ABSTRACT: The Cork Harbour area of southwest Ireland lies within the late Carboniferous Variscan Orogenic Belt. The area is characterized by a series of horizontal upright east-west anticlines and synclines, the former cored by Upper Devonian sandstones and shales and the latter by massive Lower Carboniferous limestones. During the Pleistocene, glaciers preferentially eroded the weakened limestones within the synclines. As the glaciers retreated at the end of the Pleistocene and sea level rose, deep buried valleys were infilled by glaciofluvial sands and gravels. These buried valley aquifer systems represent one of the major sources of groundwater in the area. Site investigation boreholes indicate a width of the order of 500−750 m and a depth of at least 60 m for one of the buried valleys, which can be traced for at least 60 km in an east-westerly direction. Granulometric analyses of gravel samples from boreholes determined hydraulic conductivity (K_f) of approximately 7×10^{-3} m/s. Pump tests on one borehole indicate that individual wells can give yields in excess of 20 l/s. Apart from the groundwater resources, shallow groundwater in these buried valley aquifers is locally also a heat resource due to the 'heat island' effect and can be exploited for space heating/cooling purposes.

INTRODUCTION

Cork, the second largest city in Ireland with a population approaching 250,000 within its environs, is situated in the southwest of Ireland, at the mouth of the River Lee. The Lee drains into Cork Harbour, a glacially eroded, almost completely enclosed body of water, connected by a narrow entrance to the Atlantic Ocean. The area of investigation is an extensive region of about 1,000 km^2 around Cork City and Harbour (Figure 1).

Rapid industrial growth in the Cork area has led to a significant risk of groundwater pollution, particularly within shallow aquifers composed of sands and gravels infilling buried valleys. A research project to assess groundwater resources and quality within the Cork Harbour area, jointly funded by Cork County Council and Cork City Council, the two local authorities administrating the area, is currently in progress.

This paper represents a continuation of a preliminary project report presented at IAH XXXI in Munich (Allen & Milenic, 2001) and deals specifically with the evaluation of groundwater resources in buried valley ribbon aquifers within the study area, together with a proposed methodology for hydrogeological investigations for buried valley aquifers.

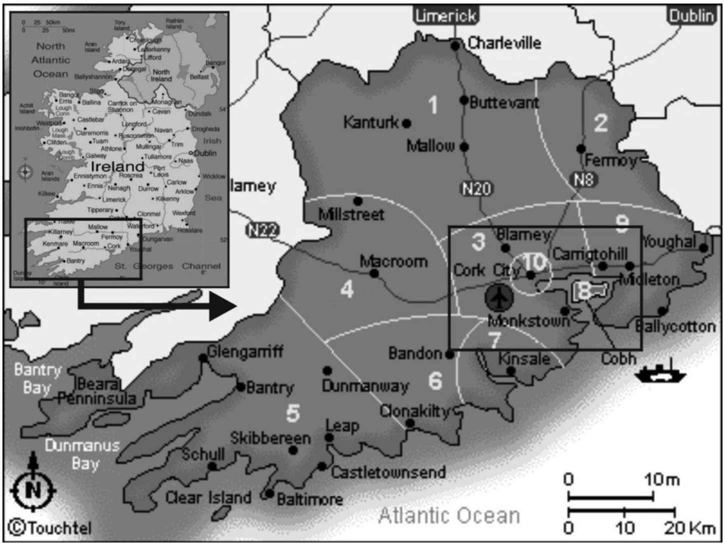

Figure 1. Location and geographical setting of the study area.

METHODOLOGY OF INVESTIGATION

Development of a basic methodology for the investigation of buried valley aquifers has stemmed from the fact that previous hydrogeological investigations in the study area were somewhat cursory, necessitating an approach from first principles and thus application of all the phases of a hydrogeological investigation. The main goals of the study have been the evaluation of available groundwater resources, water quality and the suitability for multipurpose use, and a forecast of possible changes with time. The methodology established involves the following steps (Milenic, 2004):

- collation of all geological and geomorphological background data
- determination of the meteorological and hydrological characteristics of the study area
- aquifer delineation (horizontal and vertical extent, definition of the productive zone, boundaries and aquifer geometry)
- aquifer recharge characteristics
- hydrodynamic characteristics
- aquifer discharge
- estimation of groundwater reserves
- physical properties and chemical composition of the groundwater
- evaluation of groundwater vulnerability and salt water intrusion
- assessment of groundwater uses
- low enthalpy geothermal energy ground water uses due to heat island effect.

To obtain solutions to the above tasks, a diverse series of geological and non-geological techniques have been employed:

- utilization of GIS in development of hydrogeological and digital elevation models
- isotopic analyses (^{18}O, ^{2}H and ^{3}H) and correlation with the local meteoric line

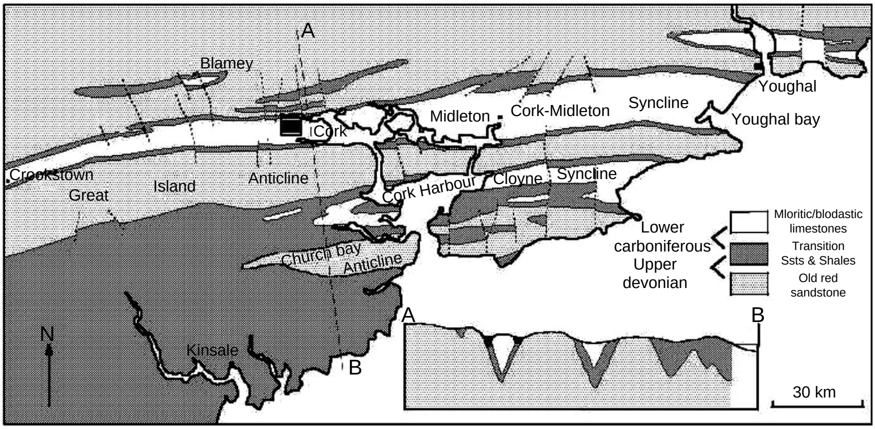

Figure 2. Simplified geology of the study area (Allen et al., 1999).

- various types of pumping test
- variable-density groundwater modelling
- groundwater balance methods
- tracing methods within the unsaturated zone and effective infiltration determination
- determination of the heat island effect in groundwater below Cork City
- optimization of potential fields of groundwater exploitation, etc.

GEOLOGY AND GEOMORPHOLOGY

The Cork Harbour area of southwest Ireland lies within the very low grade Rheno-Hercynian fold-thrust terrain of the late Carboniferous Variscan Orogenic Belt. The area is characterised by a series of horizontal upright east-west anticlines and synclines, the former cored by Upper Devonian sandstones and shales and the latter by massive Lower Carboniferous reef limestones (Figure 2). The folded sequence is cut by east-west thrusts and steep north-south compartmental faults. Tropical conditions during the Tertiary resulted in erosion of Mesozoic cover, intense karstification of the highly fractured Carboniferous limestones and the establishment of a north-south drainage pattern.

The topography of the region is controlled by the geological structure, with the anticlines forming upland areas and the synclines occupied by valleys (Figure 3). These valleys were formed during the Pleistocene glaciations (2 Ma–10 ka), as prior to this, during the Tertiary period, the regional topography sloped southwards and the region was drained by southerly flowing rivers. This Tertiary drainage was truncated by glaciers advancing outwards from the mountainous regions of western Ireland, preferentially exploiting the weaker shales or karstified limestone coring the synclines, resulting in the development of a number of broad u-shaped valleys, where previously there had been only north-south drainage patterns. Superimposed on these u-shaped valleys are a number of buried valleys infilled with sand and gravel. At the peak of the last glaciation 15,000 years ago, when much of Europe and North America was covered by ice, sea

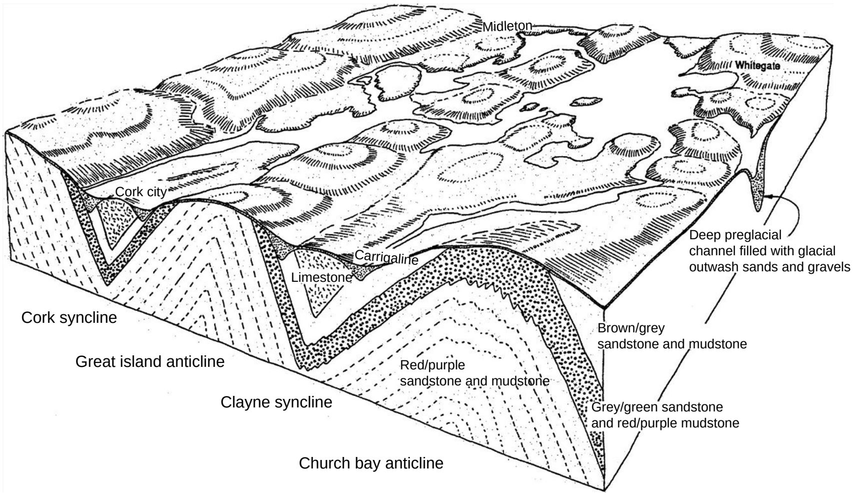

Figure 3. Block diagram illustrating the topography and structure of the Cork Harbour area (MacCarthy, 2001).

level fell to an estimated 130 m lower than at present, so the rivers eroded down to the new base level cutting deep steep-sided gorges. When temperatures subsequently ameliorated towards the end of the Pleistocene glacial epoch about 10–12,000 years ago, the ice sheets receded, sea level rose again and the gorges rapidly became infilled with fluvioglacial sands and gravels as the rivers responded again to changing base level. Furthermore, the south of Ireland is still sinking, so sea level continues to rise by an estimated 16 m over the last 8,000 years.

GENERAL HYDROGEOLOGY

Major aquifers in the Cork Harbour area occur in both bedrock and overburden deposits.

The main bedrock aquifers are the intensely karstified limestones, coring the Cork-Midleton and Cloyne Synclines, which possess significant storage capacity and hydraulic conductivity. These limestones represent typical buried karst with a surface outcrop of only 5 per cent. Effective porosity is conservatively estimated at 1 per cent with the depth of karstification of at least 60–100 m. Characteristic well yields are in the range from 3 to 20 l/s (Sleeman & Pracht, 1994).

Intergranular aquifers overlie the limestones and are developed in sands and gravel infilling the buried valleys. These are discussed in detail in subsequent sections.

The above aquifer types are in hydraulic connection.

In general, the Old Red Sandstone represents a hydrogeological barrier on a regional scale. Locally, however, a certain amount of groundwater may be contained within the weathered zone to an approximate depth of 10 m. Effective porosity is estimated at 0.1 per cent. Furthermore, north-south fracture zones cutting the Old Red Sandstone may act as pathways for groundwater transfer between the two karst aquifers within the adjacent synclines.

Table 1. Mean annual values for all analysed climatic factors recorded at Cork Airport meteorological station over the period 1963–2000 (Allen & Milenic, 2001).

Factor	Unit	Value
Precipitation (P)	mm	1,222
Temperature	°C	9.6
Relative humidity	%	85.7
Atmospheric pressure	hPa	1,115
Wind speed	knots/dir.	11–16/SW
Evapotranspiration (E)	mm	486
Soil moisture (ΔW)	mm	13

WATER CYCLE IN THE CORK CITY AND HARBOUR AREA

Major factors in shaping the climate of the Cork City and Harbour area are the predominantly westerly atmospheric circulation of the middle latitudes and the Atlantic Ocean, which lies to the north, west and south of Ireland. The maritime influence is strongest near the Atlantic coast and decreases with distance inland. For essential calculations of the major climatological parameters for the project, data for the last 37 years were used (Table 1).

Surface run-off (Q) changes with topography and is estimated at 10 per cent in the synclines (0–30 m above sea level – mASL) and 75 per cent in the anticlines (0–196 mASL), respectively.

For estimation of effective infiltration ($i_{ef.}$), the general water balance equation has been used:

$$i_{ef.} = P - (E + Q + \Delta W)$$

Within the buried valleys, the calculated value for effective infiltration is:

$$i_{ef.} = 650\,\text{mm/year}$$

BURIED VALLEY RIBBON AQUIFERS

Taking the Lee Buried Valley as a case study for the purposes of this presentation and proceeding according to the methodology outlined above, evaluation of groundwater resources is described in the following subsections.

Aquifer delineation

The lateral extent of several buried valleys in the Cork Harbour area can be delineated for tens of kilometres. A digital elevation model constructed for this purpose illustrates the main locations of these east-west trending valleys (Figure 4).

A typical example is the Lee Buried Valley (1 on Figure 4) shows a lateral extent of at least 60 km (Allen et al., 1999). It underlies the Cork-Midleton Syncline for all of this distance, but underlies the present course of the River Lee for only 10 km, from where the latter enters the Cork-Midleton Syncline at Inniscarra eastwards to Dunkettle where it

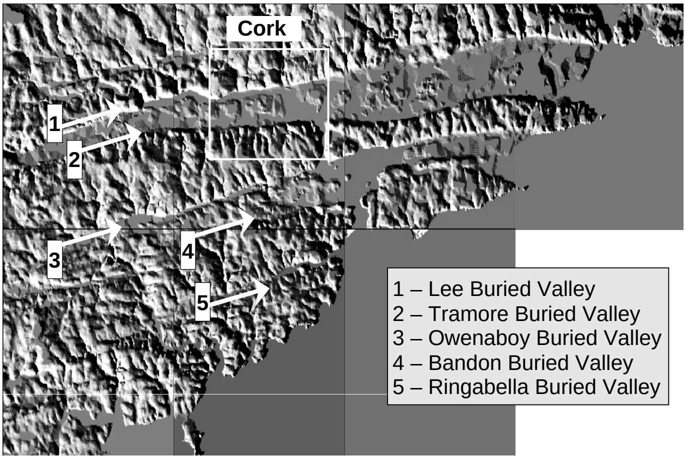

Figure 4. Digital elevation model of the greater Cork Harbour area illustrating the locations and courses of the east-west buried valleys.

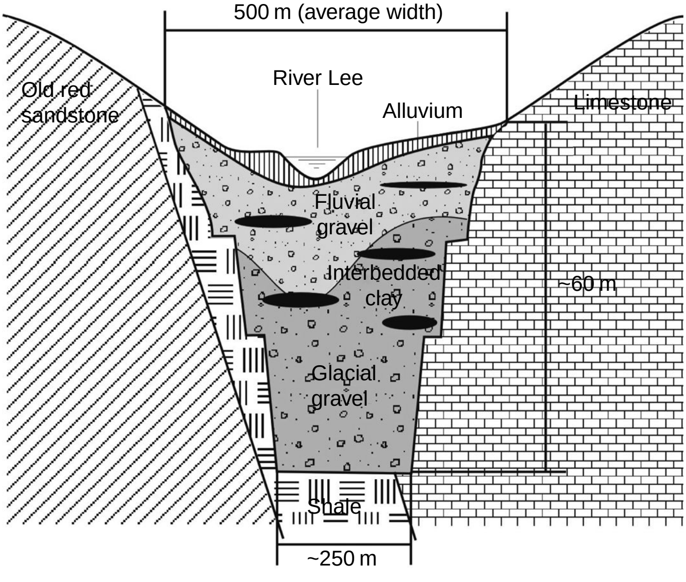

Figure 5. Schematic cross-section through the Lee Buried Valley (Allen & Milenic, 2003).

turns south to enter Cork Harbour. Where the Lee Buried Valley underlies Cork City, an abundance of bedrock outcrops and site investigation borehole data made it possible to delineate its extent fairly precisely and it is approximately 0.75 km wide. Although most site investigation boreholes extend no deeper than 20 m, a few boreholes drilled to depths of up to almost 50 m, without encountering bedrock, indicate that beneath Cork the buried valley extends to at least 50 m depth. At Carrigtohill, 10 km to the east of Cork, a borehole encountered bedrock at 60 m indicating a depth of at least 60 m for the Lee Buried Valley. A schematic profile of the Lee Buried Valley is presented in Figure 5.

In the light of the estimated geometry, the volume of the aquifer is calculated as:

$$V_{aquifer} = 375 \times 10^6\,m^3$$

Aquifer recharge

On the basis of a detailed analysis, it is concluded that the aquifer is undergoing recharge mainly through the following processes:

- direct infiltration of rainfall in areas of open gravel extraction pits, in the western part of the study area
- percolation through overlying alluvial clay sediments, ranging in thickness up to 4–5 m
- hydraulic connection with various surface streams including the River Lee, brackish within tidally influenced areas adjacent to Cork Harbour
- hydraulic connection with the surrounding karst aquifer within the Cork Syncline.

The aquifer is recharging by direct infiltration over an area of about 2 km^2, with an amount of approximately $1.3 \times 10^6\,m^3$/year.

Percolation, which has been determined as the drainage through a 5 m-thick, fully saturated layer of clay in flat terrain, is within the range of 0.05–0.3 cm/h depending on the intensity and duration of precipitation. Aquifer recharge by this process over an area of 27 km^2 represents an amount of up to $2.6 \times 10^6\,m^3$/year.

The potential total groundwater replenishment by fresh water through direct recharge and percolation, has been calculated to be approximately 19 years.

Analyses of the other recharge parameters is still in progress, so these results cannot be included in this presentation.

Hydrodynamic characteristics

Hydraulic conductivity (k) and transmissivity (T) were calculated using particle size distribution methods from borehole samples and by pumping tests on wells and an infiltration gallery.

For the particle size distribution, the USBR method was used, providing the following mean value for k (Milenic & Allen, 2001a):

$$k = 6.87 \times 10^{-3}\,m/s$$

Pumping tests were carried out on several horizontal and vertical water abstraction structures, from which representative examples are emphasized: i) an infiltration gallery at Cork Waterworks (Figure 6); and ii) a well near Cork County Hall, which captures the upper levels of the aquifer. For the k and T calculation, the recovery level method ($s = f(\log t/t - t_1)$) was used as the most reliable for these examples (Figure 7). Characteristic values are in the range (Milenic & Allen, 2001b):

$$k = 4.0-4.8 \times 10^{-3}\,m/s$$

$$T = 1.22-1.43 \times 10^{-2}\,m^2/s$$

Comparative analysis shows almost perfect correlation of results by the various methods. Therefore, these values can be considered as representative.

Effective porosity for the sand and gravel is around 25 per cent. The porous media is homogeneous in horizontal extent with a slight vertical heterogeneity and zonality.

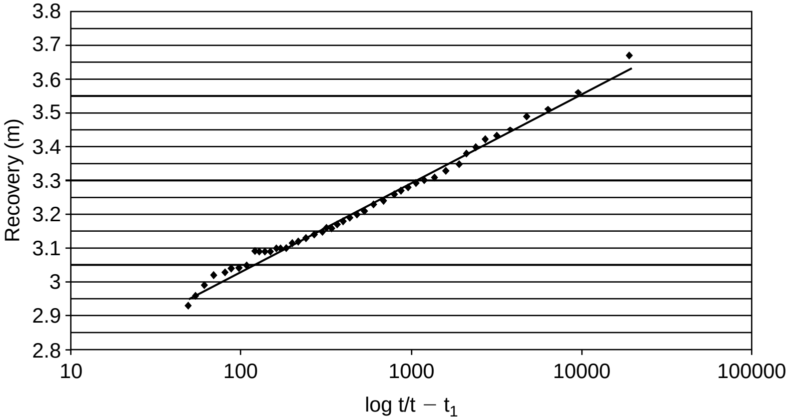

Figure 6. Water recovery data for the calculation method $s = f(\log t/t - t_1)$ (Milenic & Allen, 2001b).

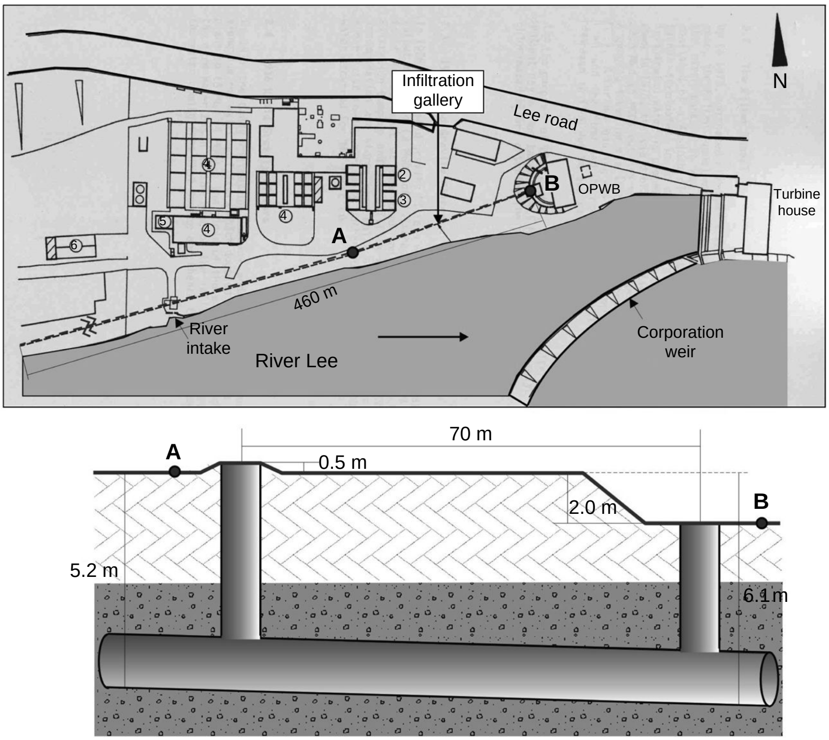

Figure 7. Position and technical details of the infiltration gallery (Milenic & Allen, 2001b).

The aquifer is characterized by subartesian pressure. The general groundwater direction is east-west, typified by a steady flow gradient (10 m of head difference for water levels of 10 km in length) and a characteristic laminar flow regime.

The static water level is close to the surface and, adjacent to Cork Harbour, is tidally influenced ranging from −1.8 m at high tide to −3.8 m at tidal lows.

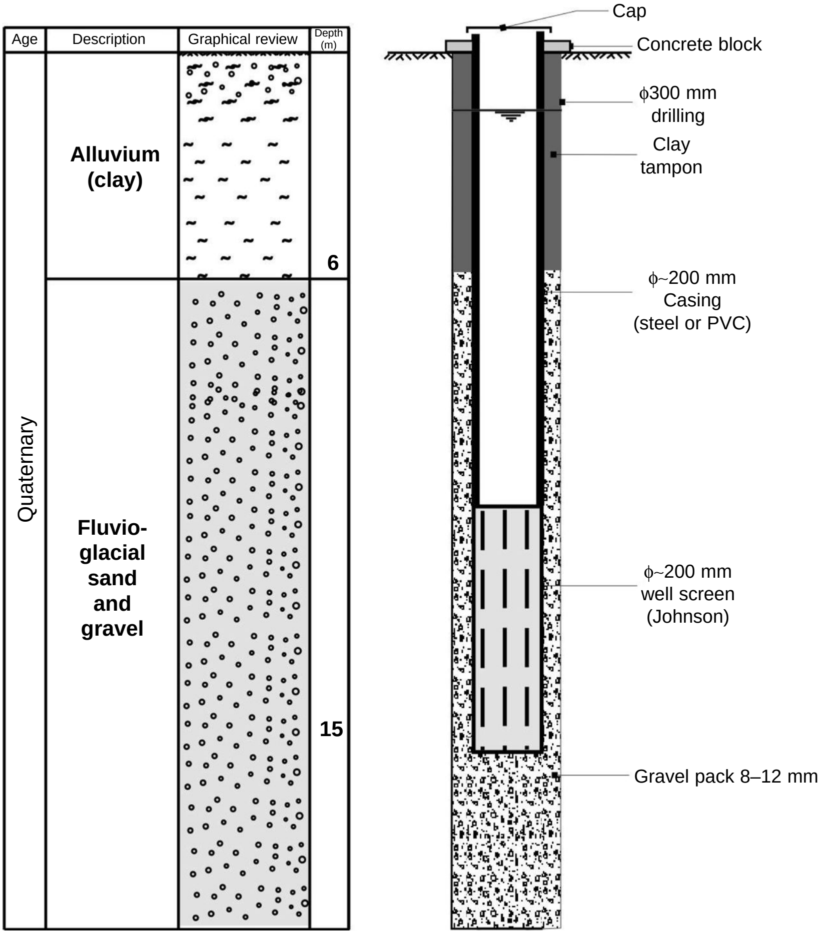

Figure 8. Characteristic depth, thickness, lithological profiles and well construction elements in wells from the study area.

Aquifer discharge

No natural occurrences of groundwater discharge have been observed in the study area (submarine areas are not included in this analysis – the possibility of submarine discharges will be subsequently evaluated using infrared satelite imagery techniques).

Aquifer discharge takes place artificially through a number of dug and drilled wells, and also through the infiltration gallery located at Cork Waterworks. The latter is of particular interest since it was constructed in 1879 and is still in operation (Figure 6). At the time of construction, the capacity of the gallery was around 260 l/s, whilst nowadays capacity has been reduced to 50 l/s, mainly due to clogging of the brick screen.

Typical well constructions have diameters of 150 or 200 mm, depths of up to 20 m and frequent use of Johnson screens (Figure 8). The capacities of the wells are up to 20 l/s, for a drawdown of around 7 m. This is probably not a maximum well yield, but a consequence of defectiveness in aquifer capture (instead of capturing the last third of the thickness of the intergranular aquifer, most wells capture the upper levels), limitations in well-casing diameter and the inability to utilize stronger submersible pumps, etc.

Preliminary calculations indicate linear well disposition as optimal for water abstraction (at every 75 m within the study area), with a transverse monitoring network (every 50 m).

Groundwater reserves evaluation

In the light of the presented data, static groundwater reserves were evaluated using classical groundwater balance equations. Since the groundwater level corresponds with sea level (which is the erosional base for Cork Harbour), the dynamic component of the reserves above the erosional level was not calculated.

Thus, groundwater reserves formed within the ribbon aquifer of the Lee Buried Valley sands and gravels are at an absolute minimum:

$$V_{GWmin} = 75 \times 10^6 \, m^3$$

This value is calculated for an aquifer length of 60 km, a productive width of 250 m, a thickness of the productive zone of 25 m and a minimum effective porosity of 20 per cent.

Physical properties and chemical composition of the groundwater

Analysed groundwaters from the Lee Buried Valley aquifer are predominantly of the HCO_3-Ca type, with typical mineralization of up to 500 mg/l, except within the zone of tidal influence where mineralization rises to 1000 mg/l and pH values to 8.5. A typical analysis is shown in Figure 9.

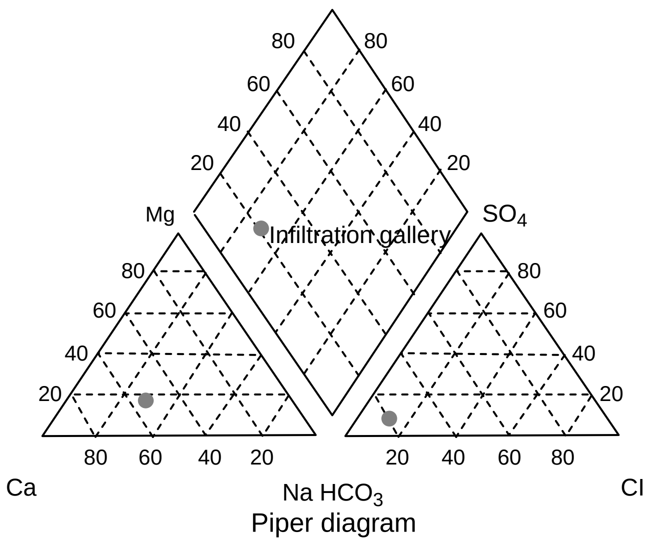

Figure 9. Piper diagram of the chemical composition of groundwater from the infiltration gallery.

Temperatures of the groundwater vary from 9–13°C. More details of the temperature properties of the aquifer are discussed in the section on geothermal energy.

Possible chemical limitations to groundwater use are reflected in significant contents of manganese and iron (from the rock matrix) and seasonal variations in nitrate concentrations due to intensive fertilizer use in agriculture, as well as bacteriological contamination in certain samples, particularly from urban areas.

The groundwater has a meteoric origin, which is confirmed by isotopic analyses of a number of samples. For this purpose, a local Cork meteoric line was constructed and then compared with the contents in the analysed samples of the natural isotopes δ^2H and $\delta^{18}O$, related to Standard Mean Ocean Water (SMOW) (Figure 10).

Groundwater vulnerability

As mentioned earlier, rapid industrial growth in the Cork area has led to an increased risk of groundwater pollution. The Geological Survey of Ireland (GSI), in cooperation with the Irish EPA and the Department of the Environment, has developed a groundwater protection scheme methodology (Kelly & Wright, 2000). Currently, the GSI is in the process of producing a groundwater vulnerability map for the Cork region on a 1:50,000 scale.

In parallel with these developments, it is necessary to identify the conditions, potential and sources of pollution on a larger scale. These sources are numerous (see Figure 11) and due to this fact a database of potential and actual polluters has been created. These include amongst others:

- chemical industries
- pharmaceutical industries
- oil refinery
- harbour
- municipal infrastructure
- active and decommissioned landfills
- old sewage pipelines
- salt water intrusion.

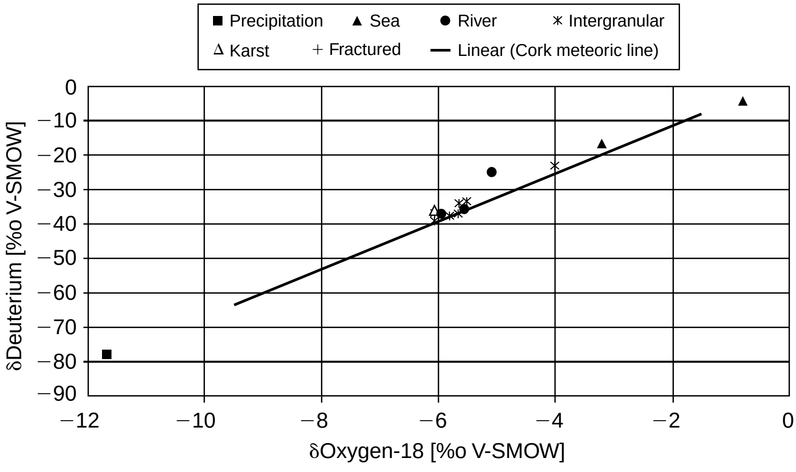

Figure 10. Isotopic composition of water in the study area.

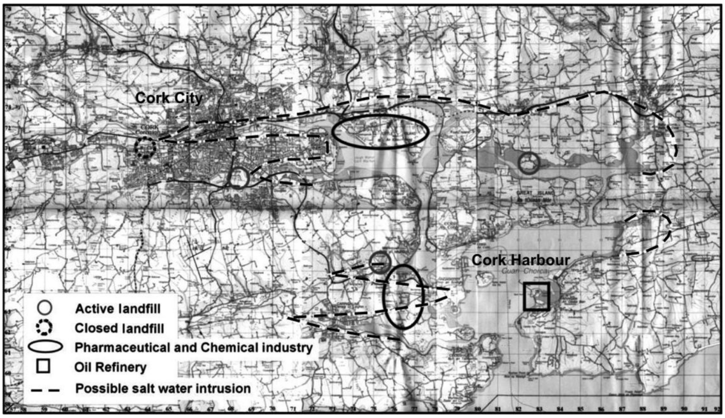

Figure 11. Main sources of pollution in the study area.

The Lee Buried Valley Aquifer is particularly susceptible to salt water intrusion due to the low-lying character of the Cork-Midleton Syncline and the extension of the Inner Cork Harbour into it. The River Lee is tidal through central Cork City and all the wells in the Lee Buried Valley Aquifer within Cork City show tidal variation in water level and mineralisation content. A salt-water intrusion model based on variable-density modelling is currently being constructed, using Argos ONE and USGS SUTRA Gui 3D code. Water level monitoring and chemical and isotopic analyses indicate that the salt-water front may extend for up to 5 km inland and thus may cause significant groundwater pollution within a wider area, despite the relatively high annual precipitation of the Cork Harbour area.

Groundwater use

Although the studied groundwater in the Lee Buried Valley Aquifer possesses enormous potential, use of this resource is still largely under-utilized. The present water supply system for the Cork City and Harbour area is based on surface water processing, using River Lee water either from Inniscara Reservoir or from an intake point at Cork Waterworks.

Only a few industrial plants use groundwater, mostly for heating/cooling purposes, such as Midleton Distillery near Cork Harbour, the largest distillery in the Irish Republic.

Thus, the main aspects of groundwater use within the Cork Harbour area can be catalogued in just a few points:

- groundwater use for industrial purposes in the chemical and pharmaceutical industries
- groundwater use for space heating/cooling purposes
- groundwater use for local water supply schemes.

Low enthalpy geothermal energy use from groundwater

The Lee Buried Valley aquifer contains significant shallow level groundwater resources, which exhibit slightly enhanced temperatures below Cork City and so are potentially exploitable as a space heating/cooling resource. This phenomenon is due to the 'heat island' effect, whereby urbanization gives rise to microclimatic changes resulting from

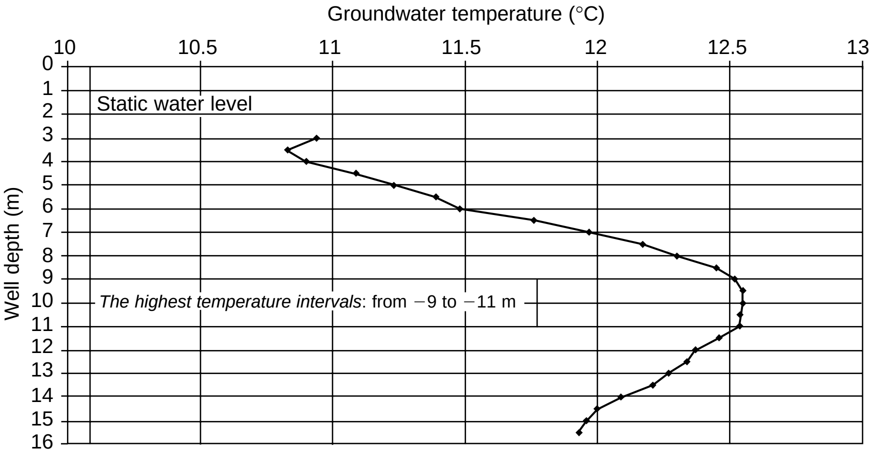

Figure 12. Vertical temperature profile determined in a well in central Cork City.

the replacement of natural ground surfaces by those characteristic of a city. This arises from the generation and trapping of heat by a combination of causes, including urban pollution 'domes', trapping of long-wave radiation beneath the urban canopy, the high thermal absorption of concrete and tarmac surfaces, and anthropogenic heat due to population concentrations in cities and conurbations (Allen et al., 1999). In Cork City, groundwater temperatures rise from 9°C (outside the city) to 13°C in the city centre. A recently determined vertical temperature profile is illustrated in Figure 12.

The nominal available geothermal energy resource (E_G) in kilowatts (kw) can be calculated using the equation:

$$E_G = H \times F \times \Delta T$$

where H = heat energy generated (kJ), F = flux of water (well yield – l/s^{-1}) and ΔT = temperature reduction in the heat pump (°C).

Based on an energy generation of $4.2\,kJ\,l^{-1}\,°C^{-1}$, a well yield of $20\,l/s^{-1}$, urban groundwater temperatures of 13°C and a temperature reduction in the heat pump of 8°C, a nominal geothermal heating resource of 672 kw may be available from wells within urban areas. This can supply space heating for buildings with a footprint in excess of $11{,}000\,m^2$.

CONCLUSION

A methodology for the hydrogeological investigation and evaluation of groundwater resources within ribbon aquifers defined by buried valleys has not been adequately described before. Reasons for this are due to the relatively limited importance of these aquifers on a regional scale and to certain similarities to the methodology of investigation of alluvial plain aquifers.

This paper presents a complete investigation of the groundwater regime and resources of the Lee Buried Valley Aquifer, providing a background for further detailed hydrogeological investigations.

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CHAPTER 15

The study of an irrigation/drainage system in a semi-arid region

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ABSTRACT: Choele Choel Island in semi-arid Patagonia, Argentina is bordered by two branches of the River Negro. Its intensive fruit production is sustained by an irrigation/drainage system. Despite improvements in crop productivity since the inception of irrigation, some portions of the island are still affected by water table build-ups. Both branches of the river provide about 155 km of aquifer-stream contact to drain groundwater. Moreover, about 73 km of artificial drains contribute to the evacuation of excess water, which is discharged in both river branches at locations downstream. Nevertheless, high river levels, caused by hydroelectric power generation from upstream dams during the peak of the irrigation season, create backwater effects at discharge points. Farmers contend that electricity power companies trigger the problem, while the power companies claim that inefficient irrigation practices are the actual cause. This work describes the results of a numerical model used to investigate the behaviour of the island's unconfined aquifer. Primary objectives of the study were to describe the relevant characteristics of the whole system during a complete irrigation period and to test the effectiveness of corrective measures aimed at reducing water table build-ups in critical areas.

INTRODUCTION

Irrigation is used all around the world to either improve agricultural production or allow crop growth in semi-arid regions. Drain tiles and canals are constructed in irrigation fields to remove water and evapoconcentrated salts from the root zone to maintain a suitable crop-growing environment. In brief, water table build-ups, subsurface flow, overland flow, stream flow, water surface storage, evapotranspiration and soil salinization are just some of the processes involved in drainage and irrigation systems.

Choele Choel Island, bordered by two branches of the River Negro in semi-arid Argentinean Patagonia, is one of the main fruit production regions of the country. With an annual average rainfall of 150 mm, any agricultural activity in the region can only be sustained thanks to an irrigation/drainage system. In spite of great improvements in crop productivity since the inception of the irrigation system, some portions of the island are

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still affected by poorly drained, high water table soils. During the irrigation season, losses through unlined distribution canals and in irrigated fields cause severe water table build-ups in some locations, jeopardizing the productivity of fruit plants and creating soil salinization problems due to the evaporation of groundwater.

Along its upper valley, the River Negro is largely controlled by a series of dams. In the middle valley, where Choele Choel Island is located, the river bifurcates into two branches. The north branch conducts more than 90 per cent of the river streamflow, while the south branch and the derivation channel the remaining flow (Figure 1). Both river branches act as natural drainage canals, providing over 155 km of aquifer-stream contact along the perimeter of the island. Drain tiles, ditches and main collectors add 73 km of an artificial network to evacuate excess water, which is discharged in both branches at locations downstream. Nevertheless, high stream stages, caused by hydroelectric power generation from upstream dams during the peak of the irrigation season, create a plug flow situation and backwater effects at main collector discharge points. Farmers contend that water table build-ups are triggered by hydroelectric power companies due to high dam releases, while the power companies claim that inefficient irrigation practices are the actual cause of the problem.

Numerical modelling is a commonly used tool to address various issues regarding drainage/irrigation systems (see for example the work of El-Sadek et al., 2001; Kupper et al., 2000; Manguerra & García, 1997). The River Negro State Water Authority perceived the usefulness of regional groundwater modelling as a valuable tool for getting a better understanding of the system as a whole, as well as for testing some corrective measures in order to improve poor drainage conditions.

This work describes the use of the groundwater model MODFLOW to investigate the unconfined groundwater system on the island and its interaction with surface sources. The primary objectives of this study were to describe the relevant characteristics of the

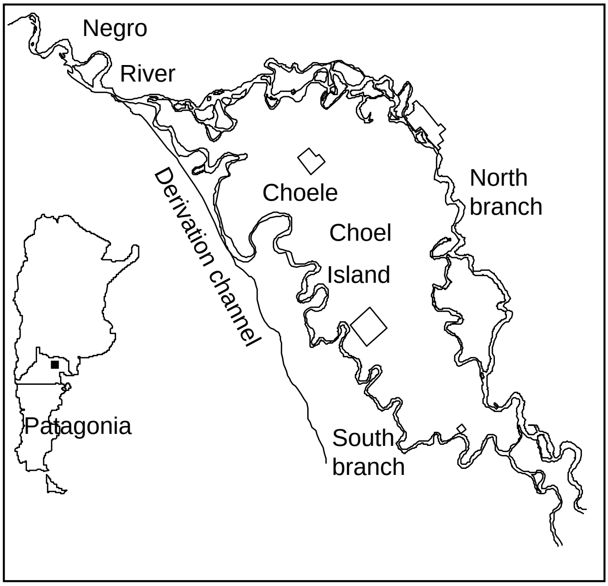

Figure 1. Location of the study area.

system during a complete irrigation period and to test the effectiveness of some correction measures aimed at reducing water table build-ups in critical areas.

STUDY AREA

Choele Choel Island is located in the middle valley of the River Negro in Argentinean Patagonia (Figure 1). It is bordered by the north and the south branches of the river and covers approximately 34,000 ha (1 ha = 10,000 m^2).

The valley was excavated across the Patagonian plateau by the river during geologic times. A typical stratigraphic profile is conformed by: (i) Rionegrense Formation, which occupies the lowest portion of the profile, composed of alternate layers of sand, silt and clay of moderate to low permeability; this geologic unit can reach several hundred meters, although it possesses no hydrogeologic interest; (ii) Patagonian boulders (Rodados Patagónicos), composed of conglomerates and boulders, usually cemented with calcium carbonate, its thickness reaches 3–5 m and extents all over the valley; (iii) Modern Terrace (Relleno Moderno), which occupies the uppermost portion of the profile; it is mainly composed of a thin layer of silty sand, underlaid by gravel deposits with variable quantities of sand, characterized by high permeability. The Modern Terrace is therefore the main water-bearing unit. Unconfined in most of the area, it is highly dynamic and interacts closely with all surface water sources. Transmissivity values for this aquifer range from 300–2500 m^2/day while the reported specific yield varies from 0.01–0.2. Field studies have shown that there is no hydraulic connection between the Modern Terrace and the underlying formation (Interconsul-Tahal-Ade, 1974).

Geomorphologically, the natural landscape has been disturbed because of the modifications carried out to construct the drainage/irrigation system of the island. Surface slope averages 0.58 per cent in the longitudinal direction, with the highest elevations in the northwest decreasing toward the southeast.

The irrigation system includes a main derivation channel (19.4 km), a main channel, eight secondary channels totalling 89.8 km and minor channels that reach into irrigation fields adding unlined canals of up to 61.2 km. The drainage system is composed of main collectors, canals and ditches that make up an intricate network of about 73 km.

Natural flow conditions in the River Negro have been highly modified due to the construction of power plants in the upper valley. The operations of dams cause a decrease in peak flows and an increase in low flows, creating backwater effects at the discharge points of main drainage canals.

DEVELOPMENT OF THE GROUNDWATER MODEL

MODFLOW, the well-known three-dimensional finite difference groundwater flow model (McDonald & Harbaugh, 1988), with its Stream package (Prudic, 1989) was used to simulate groundwater flow in Choele Choel Island and investigate the interaction between groundwater and surface water. The island aquifer was represented by a uniform rectangular grid of 134 rows and 64 columns (a total of 8576 cells), oriented in the direction of the regional groundwater flow. Individual cells measured 300 × 300 m. A single, 20 m layer representing the Modern Terrace Formation, was simulated. The north and south branches of the river were represented by 197 and 244 nodes, respectively (Rodríguez et al., 2001).

The single aquifer was simulated as a free-surface boundary able to fluctuate in response to recharge from irrigation fields, evapotranspiration, losses from unlined irrigation canals and interaction with the highly dynamic streams.

Flow to drains was simulated through the MODFLOW Drain package. Main drainage canals were represented by 266 cells, which closely followed the actual location of the drainage network. Losses along main and secondary irrigation channels were simulated at 349 cells. Losses along smaller distribution canals within irrigation fields were embedded in the term 'effective recharge', explained below. Figure 2 illustrates model sinks and sources as well as a detail of the model grid. The complete grid was omitted for the sake of the picture's neatness.

Evapotranspiration constitutes an important loss from the groundwater system. Luque et al. (1970) applied the Blaney-Criddle method to estimate consumption use for different crops on the island. Unfortunately information on the cultivated area is out of date, which makes current evapotranspiration estimates difficult to obtain. On the other hand, farmers use water in excess of field capacity, applying it in irrigation fields unevenly in space and time. This causes a generalized rise in the water table as well as differential water level increments. In order to simplify these complex dynamics, an effective recharge resulting from the combination of aerial recharge, recharge from minor line sources and evapotranspiration was introduced over the whole island, whose rate was allowed to vary in space and time.

The branches of the river were defined as variable head boundary conditions in the periphery of the island. In the southeastern extreme, between the north and south

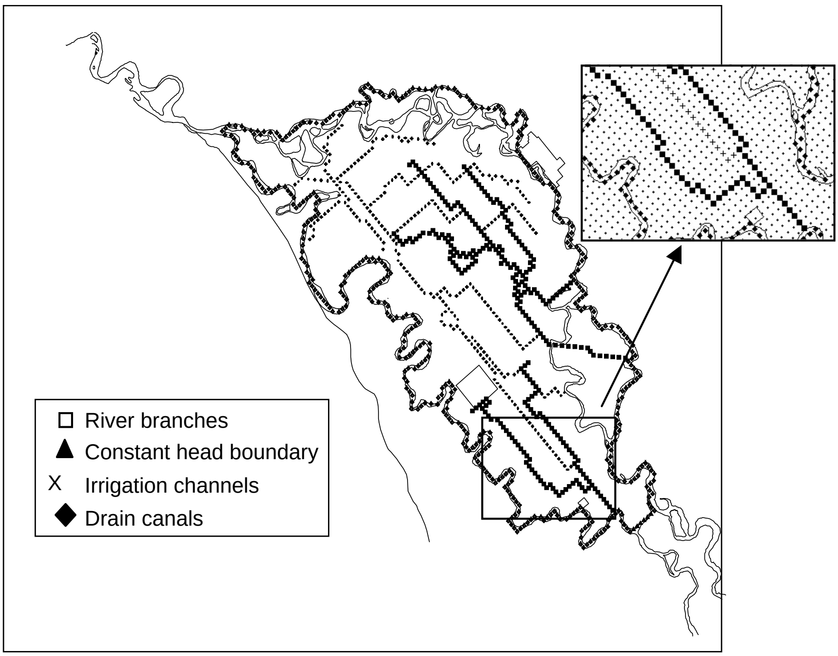

Figure 2. Model line sources, boundary conditions and detail of grid.

branches, a time variable constant head boundary condition was adopted, in accord with changing ground and surface water conditions during the irrigation season.

MODEL CALIBRATION

Model calibration refers to the process of adjusting boundary conditions and model parameters in order to get a good match between observed and simulated values of the state variables.

The groundwater model was calibrated with a trial- and-error process. Two simulations were performed: (i) a steady state simulation; and (ii) a transient simulation through a complete irrigation season. The purpose of the steady state run was two fold: on the one hand to get a set of calibrated parameters and on the other to define initial conditions for the subsequent transient simulation. The transient simulation allowed the reproduction of the temporal and spatial evolution of groundwater levels and stream stages, and established enough confidence in the model for its use as a prediction tool.

Calibration of the model was achieved by matching simulated heads with observed heads and simulated stream stages with observed stream stages. Figure 3 shows a schematic of the MODFLOW packages active on the steady state simulation along with the parameters adjusted during the calibration process.

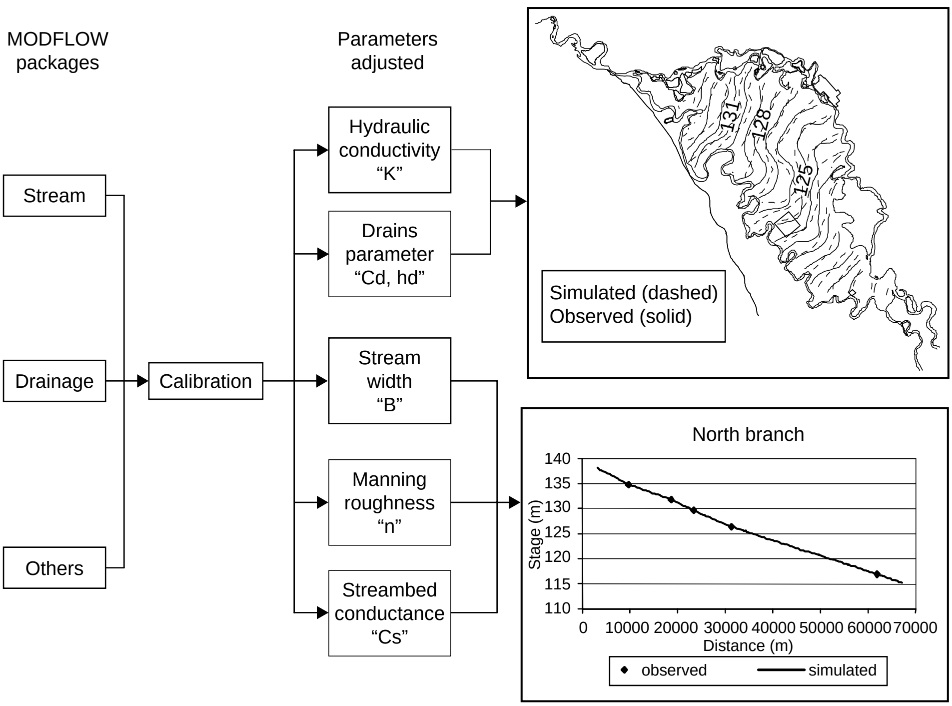

Figure 3. Steady state simulation: MODFLOW packages, calibration parameters, aquifer heads and stream stages.

Initial estimates of hydraulic conductivity 'K', Manning roughness coefficient 'n', stream width 'B' and drains stage 'hd' were either extracted from previously published studies conducted on the island (Interconsul-Tahal-Ade, 1974) or from field data supplied by the State Water Authority. Values for drains conductance 'Cd' and streambed conductance 'Cs' were adjusted during calibration as no previous information existed.

Biweekly water table elevations were available at 118 piezometers over the island, i.e. an information density of 1 piezometer every 257 ha, considered adequate for the study objectives.

Stream-related data were not as consistent as aquifer data. Stream flows from an upstream gaging station located on the River Negro were distributed between the two branches and the derivation channel by means of a partition coefficient. Approximately 90 per cent of the flow was derived from the north branch. Stage values used for calibration were available at four or five points along each branch; however, this information presented some deficiencies, a fact that caused certain local problems on the calibration of stream-related parameters.

STEADY STATE MODEL RESULTS

June, 1998 was defined as the steady state month. In this month most of the water accumulated in the saturated storage during the preceding irrigation season had been evacuated through drainage canals and natural streams. In addition, irrigation channels are inactive during May, June and July of each year, which, in turn, simplified the conceptual model.

Given that the maximum observed water table fluctuation throughout an irrigation/drainage period was around 3 m, the calibration target for heads was set at ± 0.3 m.

The upper right corner of Figure 3 shows simulated and observed heads on the island. Model results, characterized by smooth contour lines and regular gradients, reproduced general groundwater flow patterns fairly well. The comparison between simulated and observed heads was very good in the central and northwest areas of the island. It is in those areas where the highest water table fluctuations occur and data quality is best. The comparison was not as good near the south branch due to scarce data for comparison purposes and uncertainty about some model parameters.

Observed stage fluctuations for the south and north branches were in the order of 0.4–0.5 m and 1.4–1.5 m, respectively, for a range of River Negro discharges between 517 m^3/s and 1275 m^3/s. Therefore, the calibration target for this variable was set at ± 0.12 m. Table 1 shows the values of the statistical indicators used to evaluate the model's goodness of fit.

Table 1. Model errors (^{TM} = observed-simulated; E = average value of $|^{TM}|$).

	North branch	South branch
E (m)	0.05	0.04
Max $|^{TM}|$ (m)	0.12	0.10
Min $|^{TM}|$ (m)	0.00	0.00

Model errors in each individual cell were within the calibration target. As shown in the lower right corner of Figure 3, model results reproduced the observed longitudinal hydraulic profile quite well.

TRANSIENT SIMULATION RESULTS

The transient run covered the complete irrigation season from July 1998 to April 1999, with a monthly stress period. Values for initial head and stage were those generated through the steady state simulation. Model complexity increased as new processes were introduced, namely losses along irrigation channels, represented by the parameter 'Lc', and aerial effective recharge, represented by the parameter 'Er'.

Figure 4 illustrates a schematic of the new model structure. Some of the difficulties encountered during the transient simulation were related not only to the number of parameters to be handled at one time, but also to the amount of field data available. In particular, water application in irrigated fields does not occur simultaneously over the whole modelled area, but rather quite unevenly depending on the extension of farms and crop type. This problem was somehow overcome by the use of an effective recharge rate. Estimates for consumptive use, irrigation channels losses and recharge were integrated to

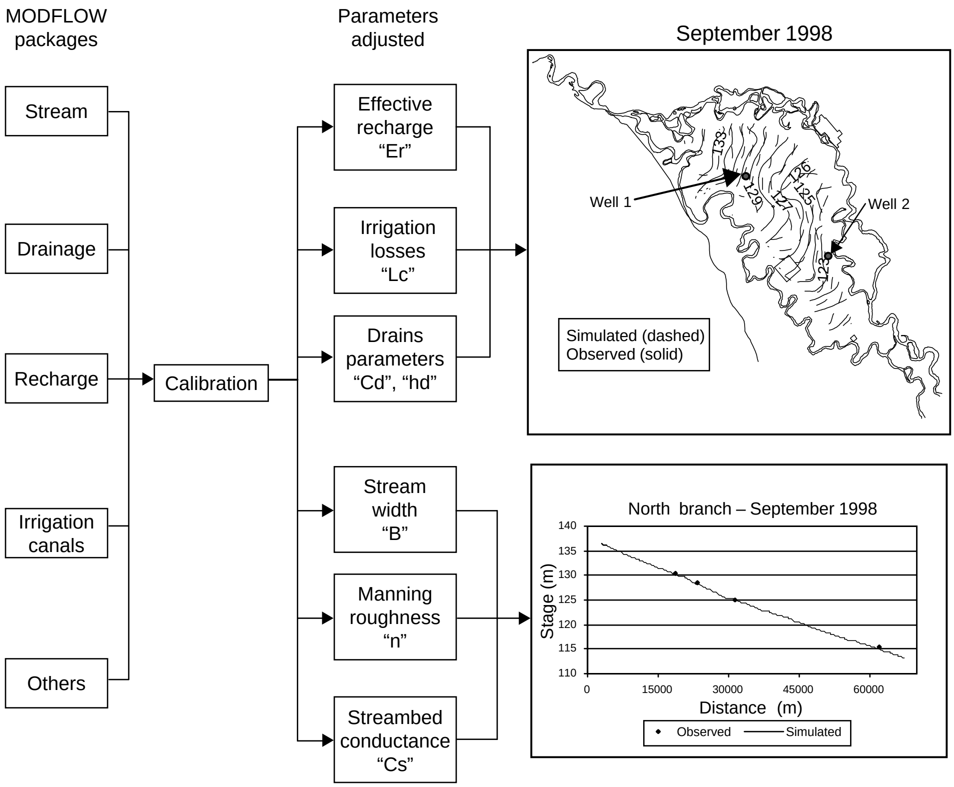

Figure 4. Transient simulation: MODFLOW packages, calibration parameters, selected results.

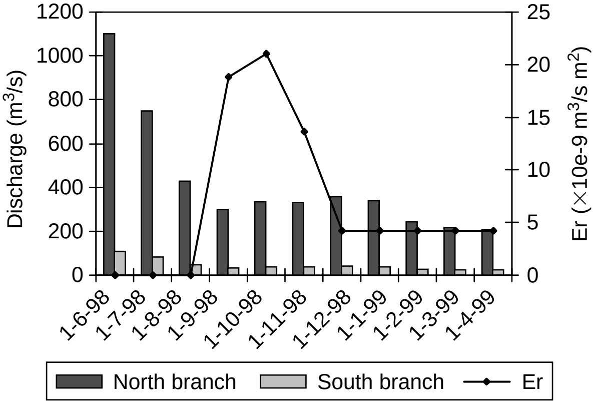

Figure 5. Transient simulation input streamflows and calibrated effective recharge.

define a first approximation for this parameter, later adjusted during the calibration process. Figure 5 shows monthly streamflows. Except for the first few months of the hydrograph, discharges stayed well below historical mean streamflows. A calibrated effective recharge rate in a representative cell has also been included in Figure 5. It can be seen that the maximum rate was applied in September and later reduced to a more or less constant value.

Calibration targets for heads and stages were equal to those set for steady state calculations. The upper right corner of Figure 4 shows heads for September 1998. The model was able to correctly reproduce general groundwater flow patterns not only for September, but also for all months. Contour lines from observed data presented irregular shapes due to differential increments in water table elevations. Comparison between observed and simulated values was good in the central portion of the island and all along the north branch, however observed and simulated values presented some discrepancies in the surroundings of the south branch where field data were less reliable.

Results were more sensitive to effective recharge rates as this variable affected great extensions of the modelled area. On the contrary, losses along irrigation channel and drain-related parameters have a more localized effect on water table elevations.

In general, except for some short reaches where observed stages were a bit underestimated by the model, agreement was good for both branches of the river. The comparison with field data was possible only through January 1999 as no stream stage readings existed afterwards. During the months that followed (February to April 1999), streamflows were of the same order of magnitude as those registered in January 1999. Therefore, the modelling strategy was to maintain constant stream calibrated parameters (Cd, B and n) for the month of January to the end of the simulation. For the sake of brevity, only the north branch hydraulic profile for September 1998 is shown in this work (see the lower right corner of Figure 4).

In addition to the analysis of the spatial distribution of water table elevations, a temporal analysis was attempted at selected piezometers. It is known that this type of analysis should be taken with some caution as the location of observation points does not always coincide with nodes where the numerical solution is obtained. Anyway, it can be considered as an additional check to build confidence in the performance of the model.

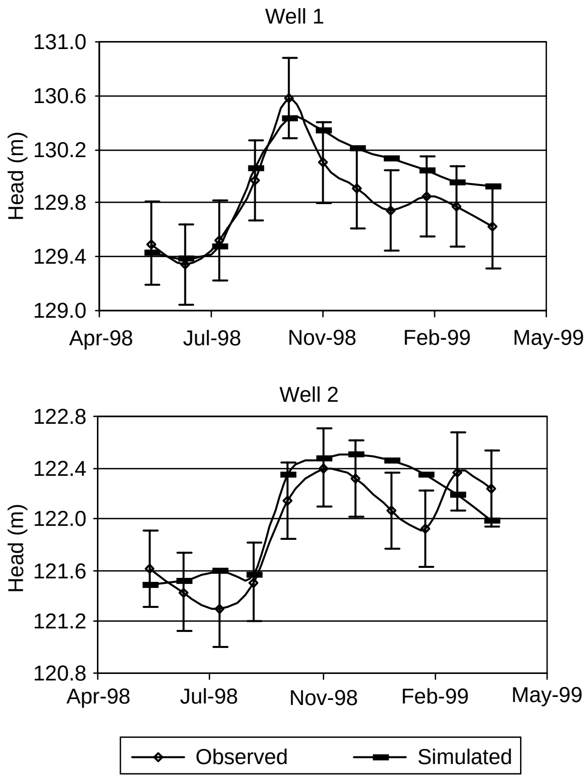

Figure 6. Simulated and observed water table elevation at selected piezometers.

Figure 6 illustrates simulated and observed water table elevations at two observation wells (see Figure 4 for their location). Water levels experience a considerable rise between August and September, in coincidence with the beginning of the irrigation season. Vertical error bars representing ± 0.30 m with respect to observed values were included to illustrate the calibration target. Volumes of water derived for irrigation are more or less constant throughout the season. However, evapotranspiration losses differ greatly from month to month, from very low values in August and April to a maximum in February (Rodríguez et al., 2001). Therefore, excess water accumulates in the phreatic aquifer causing high water levels that keep high practically until April, returning to their initial values after winter months.

MODFLOW provides a summary of fluxes through internal and external boundaries of the model. Even though field measurements for comparison purposes did not exist, some other indicators were used in order to analyse flux calculations to get a better understanding of the system as a whole. Figure 7 shows the temporal evolution of fluxes. According to the MODFLOW convention, negative fluxes represent losses from the aquifer, positive fluxes represent gains to the aquifer. Fluxes start to increase after the beginning of the irrigation season in the month of August. On the island, groundwater is drained either naturally for more than 150 km along the stream-aquifer contact or artificially through drainage canals. Model results and field indicators show that the first mechanism is more efficient, a situation favoured by low streamflows during the

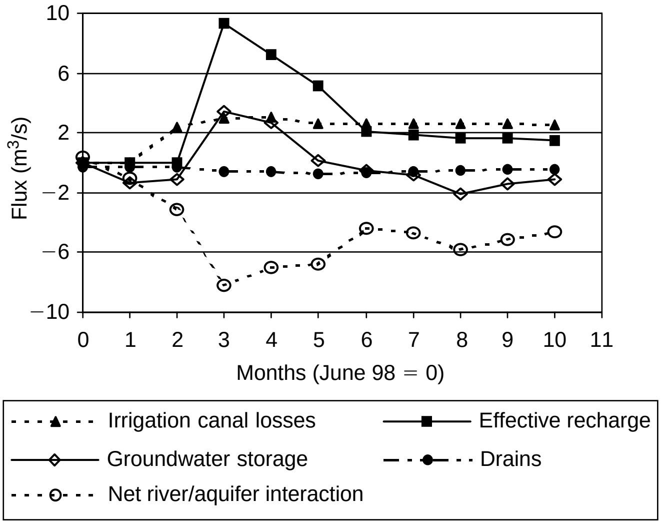

Figure 7. Volumetric budget.

simulation period, which allowed free-flowing conditions at drains discharge points. In spite of drainage, water accumulates in the saturated storage in coincidence with maximum recharge rates and later drains out towards the end of the simulation to regain its initial values.

Calibrated fluxes representing losses from irrigation channels were more or less constant from September on. The total simulated volume added to the aquifer by this source was approximately $63 \times 10^6\,m^3$. Previous estimates (Interconsul-Tahal-Ade, 1974) are of the order of $80 \times 10^6\,m^3$, a value that also includes losses through tertiary irrigation channels.

MODEL PREDICTIONS

Technical options to reduce the accumulation of groundwater in critical areas include groundwater pumping and the improvement of conveyance conditions in main drainage canals, whose current bottom elevations are up to one meter above the design elevation.

Once satisfactorily calibrated, the groundwater flow model was used as a prediction tool to test the second alternative just mentioned. Here, the interest focused on the changes experienced by the river/aquifer system in response to modifications in the bottom elevations of drains as both mechanisms help to evacuate excess groundwater. Initial conditions for the simulations were defined as those corresponding to the month with the highest water table elevations resulting from the transient simulation. This condition was met in October. Two streamflow scenarios were simulated, corresponding to average waters ($450\,m^3/s$) and high waters ($1100\,m^3/s$).

Table 2 summarizes the conditions for the simulations as well as the results obtained in terms of flux. The zero reduction bottom elevation along drain cells corresponds to current conditions. Simulations referred to as the 0.4 and 0.6 m reduction in the bottom

Table 2. Drains and stream/aquifer fluxes for prediction scenarios.

Stream flow (m^3/s)	Bottom elevation reduction (m)	Drains		Streams/aquifer interaction	
		(m^3/s)	(%)	(m^3/s)	(%)
450	0.0	−0.79	–	−7.78	–
	−0.4	−1.04	+32	−7.68	−1.3
	−0.6	−1.17	+48	−7.62	−2.0
1100	0.0	−1.02	–	−6.00	–
	−0.4	−1.20	+18	−5.94	−2.0
	−0.6	−1.34	+31	−5.87	−13.0

elevations of drains correspond to simulations where a uniform reduction of that amount was applied along all drain cells.

Results from runs with a 0.4 and 0.6 m reduction in the bottom elevations of drains were expressed in flux units of m^3/s and in percentage of flux relative to the 0.0 m reduction run. For the lowest streamflow, a 0.4 and 0.6 m reduction accomplished a 32 and 48 per cent increase in flux evacuated by drains, respectively. The stream-aquifer interaction fluxes remained practically unchanged. However, when the streamflow was increased to 1100 m^3/s, lowering the bottom elevations of drains was not as effective. Drain fluxes increased a maximum of 31 per cent for the 0.6 reduction scenario. This highlights the negative effect that high streamflows may have on artificial drainage. Natural drainage was affected as well. In this sense, notice that for the zero reduction simulation the fluxes from the aquifer to the stream reduced from 7.78 m^3/s for the lowest streamflow to 6 m^3/s for the highest streamflow, i.e. a reduction of 23 per cent in the stream/aquifer interaction fluxes.

These results are better illustrated in Figure 8, which shows changes in water table elevations for the lowest streamflow tested and a 0.6 m reduction in the bottom elevations of drains, relative to the 0.0 m reduction run for the same streamflow. The effect of this alternative is only local, influencing a small area along the drains. In the case illustrated, maximum water table declines were around 0.15 m. These water table differences are responsible for the 48 per cent increase in the drain fluxes mentioned earlier. The rest of the scenarios yielded similar results, though for the highest streamflow case, the reduction in water table elevations was of lesser magnitude.

CONCLUSIONS

A good understanding of the surface/groundwater system hydrodynamics is essential to elucidate the existing controversy between the electricity power companies and farmers on Choele Choel Island. In situations like this, numerical modelling has proved to be a very valuable tool, as shown in this contribution.

Both the steady state model and the transient model developed for this work, yielded results that adequately reproduced regional groundwater flow patterns as well as hydraulic profiles along streams. The analysis of fluxes showed that natural drainage of groundwater by streams is more efficient that drainage through manmade canals.

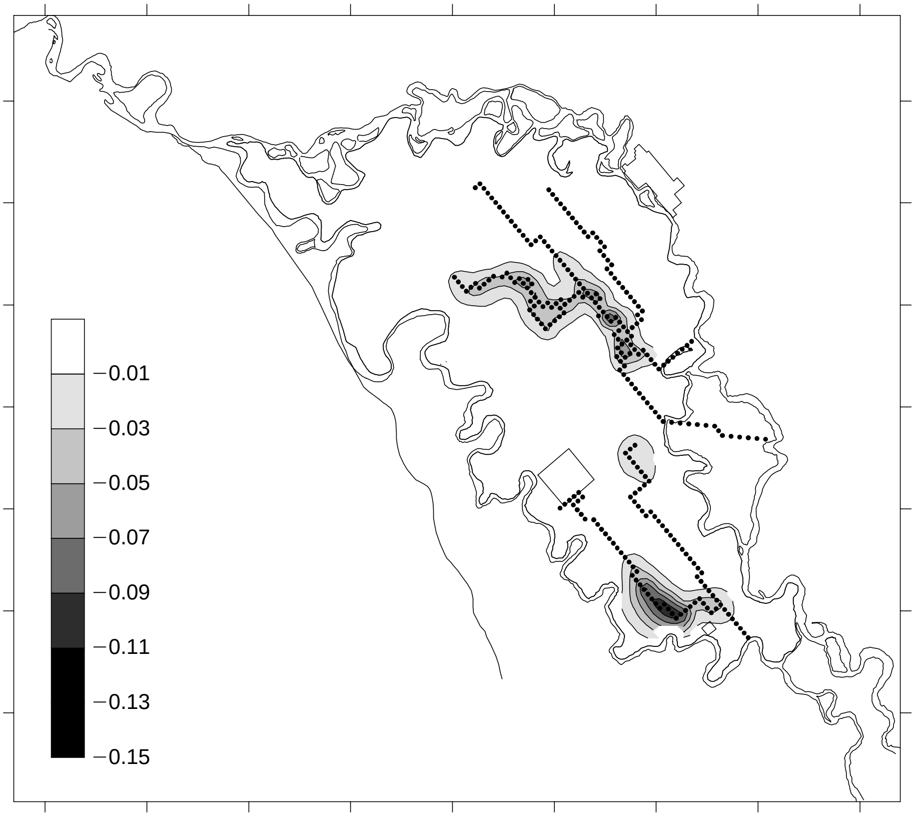

Figure 8. Effect of drainage canal bottom elevation on water table elevations ($Q = 448\,m^3/s$, 0.6 m reduction).

In the prediction stage, it was shown that, lowering the bottom elevations of drains 0.4 and 0.6 m with respect to current elevations constitutes an effective measure to increase flows evacuated by drains. However, the reduction in water table elevations is observed only along drains. On the other hand, low streamflow favoured groundwater drainage, both artificial and natural.

Management efforts should be targeted not only to structural measures but also to non-structural measures. Farmers apply water in excess of field capacity. Moreover, there is a clear indication that a great proportion of the irrigation water is not being used and hence is directly discharged into drainage canals. Therefore, reducing irrigation water allocations in combination with more efficient irrigation practices and a convenient release policy for electricity companies directed to maintain low flows during the peak of the irrigation season could substantially reduce unwelcome water table rises on the island.

ACKNOWLEDGEMENT

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CHAPTER 16

Glyphosate mobility in piedmont soils of the Australes range in the south of Buenos Aires Province

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ABSTRACT: An increase in agricultural production is leading to an intensive use of agrochemical products to improve plant growth. This study of glyphosate soil adsorption took place on an experimental plot of an agricultural sector of Buenos Aires Province. In this region the product is currently applied in areas of direct sowing and in very small doses. By carrying out batch tests the partition coefficient K_d, which relates the concentration of glyphosate in the water phase to the adsorbed one in the soil, was obtained. This coefficient was standardized according to the organic matter content (K_{oc}), and optimized using models based on the Freundlich isotherm. The greatest degree of adsorption of glyphosate occurs at surface level and decreases with depth, owing more to variations in the structure and chemical composition of the clay sediments than to the effect of the organic matter. There is a very high adsorption of the glyphosate in the soil so it falls into the category of being non-leachable. This characteristic gives it potentially little impact as a polluting agent, provided the conditions of preferential flow that could significantly increase its mobility are not generated.

INTRODUCTION

A growth in agricultural activity, due to increasing demand for high crop yields, has brought about, among other things, a deterioration in the quality of hydrologic resources. In the Bahía Blanca area and the hilly region in the south of Buenos Aires Province this issue has become particularly relevant.

Since groundwater in this area is a potential source of water supply to meet human needs, it is necessary to know and be able to quantify the soil characteristics that cause the retention of pollutants. Once the pesticides are applied to the soil surface they undergo various physical and chemical processes that govern their transit time and produce several transformations in their chemical composition. The adsorption and degradation processes are the most important; the type and content of clays in the soil are mainly responsible for the adsorption and the organic matter content for the total degradation of the pesticide and/or its metabolites, this being its main effect (Candela et al., 1998).

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Since carrying out tests for the adsorption of organic substances in the field is complex, it is advisable to perform the first characterization in the laboratory. Although less representative of the natural conditions (close system and greater specific surface of particles available for the adsorption), tests in batch are the first approach to the characterization of the pesticide distribution coefficient (K_d) between the water and the soil. The adsorption values are clearly influenced by water flow through the soil. Batch experiments may result in values that vary in several orders of magnitude with the various types of soil, depending on: (i) the physical and chemical properties of the pesticide and soil and (ii) the chemical composition of the soil solution.

In batch experiments the contaminated solution and the soil in a disaggregated state are brought into contact in a reaction vessel. After a period of time that normally ranges from a few hours to days, the degree of partitioning of the contaminant between the solution and the soil is determined. From the experimental results obtained and the distribution coefficients, K_d may be calculated by the following expression:

$$K_d = (\mu g\ \text{pesticide}/g\ \text{soil})(\mu g\ \text{pesticide}/g\ \text{water})^{-1}$$

K_d is not a constant and varies widely with the various properties of the soil studied. However, it is possible to standardize the values of this coefficient on the basis of the content of the organic matter. In this way an adsorption constant (K_{oc}) applicable to all types of soil is obtained. This constant can be calculated by:

$$K_{oc} = 100\ K_d(\text{organic carbon}\%)^{-1}$$

However most authors claim that the processes of adsorption of pesticides on soils may be best studied by using models based on the isotherm of Freundlich (K_F). This is obtained by determining the adsorption in a series of systems with different concentrations. The equation corresponding to the isotherm of Freundlich is:

$$C_s = K_F\, C_e^n$$

its linear expression being

$$\ln C_s = \ln K_F + n \ln C_e$$

where C_s is the pesticide quantity adsorbed in the soil (μg/g soil), C_e is the concentration of the water solution at equilibrium (μg/ml solution) and n is a coefficient that denotes the degree of adsorption.

In the area under study glyphosate is the most widely used pesticide. It is a non-selective herbicide, which is applied in doses not exceeding 1.5 lts/ha in sectors of direct sowing. The main objective of this paper is to evaluate the potential adsorption in soil of this compound and thus determine the risk of contamination of the piedmont phreatic aquifer of the Australes ranges.

MATERIALS AND METHODS

Location of the experimental plot

The experimental plot is located in the River Del Águila Basin, on the southwestern slope of the Australes mountain range, province of Buenos Aires, Argentine Republic (Figure 1).

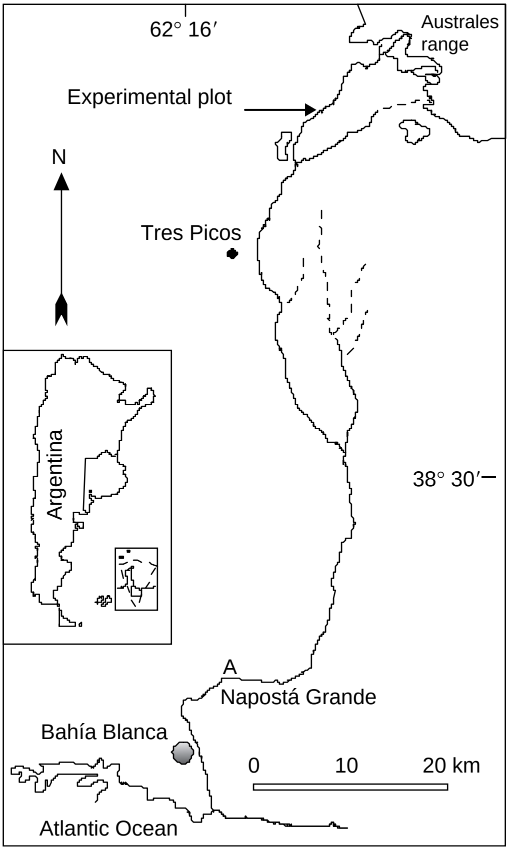

Figure 1. Location of the experimental plot.

The plot lies 70 km to the north of Bahía Blanca city, 271 m above sea level at 62° 06′ West longitude and 38° 12′ South latitude.

Physical and chemical characteristics of the soil

The soil is a Hapludol, moderately drained and located on a flat relief and gradient >1 per cent (Lexow & Bonorino, 1997). The parental material comprises loess sediments that overlay a sandy layer, probably of alluvial origin. The test for glyphosate adsorption was performed on the smaller fraction of 2 mm from three soil samples obtained in the experimental plot at 30, 60 and 90 cm of depth. Physical and chemical characteristics are shown in Table 1.

The pH value may be related to the humic acid content of the organic matter (OM); an increase in the concentration of the latter will cause an increase in the existing humic acid concentration and therefore the value of pH will be lower.

Soil hydrodynamics characteristics

The major hydrodynamic parameters show that the soils have a high total porosity (between 0.48 and 0.52); a wide range of saturated permeability with a mean 0.3 m/d; the soil characteristic curves show high storage and moisture retention conditions (Lexow & Bonorino, 1997). In the experimental plot the soil hydrodynamics characterization involved the development of diverse experimental studies. Among others a test of infiltration with a conservative tracer allowed observation of the evolution of its leaching through the soil (Lexow et al., 1998). These studies showed the existence of both matrix and preferential water flow. The vertical displacement of a non-reactive tracer with respect to time is usually measured by the change in the position of tracer peaks (maximum concentration). In a non-dispersive system the tracer will move progressively to lower depths (piston flow) without significant vertical spread of the front of the pulse. The higher the dispersion, the larger the vertical spread of tracer and larger the deviation from piston-type flow (Sharma, 1989). It was by this means that unitary rates of matrix flow from 0.8–1.6 mm/d and of preferential flow from 0.9–3.3 mm/d were determined (Lexow, 2002).

Experimental procedure

Water samples with 0, 500, 1000, 2000, 5000 and 10,000 μg/l of glyphosate concentrations (C) were prepared. Aliquots of 50 ml were taken from C, mixed with 20 g of soil and agitated for 72 hours. The supernatant solution was separated by decantation and centrifugation and the glyphosate concentration (C_e) was determined. The concentration of glyphosate adsorbed in the soil (C_s) was calculated from C minus C_e (Table 2). Glyphosate concentration, in both the soil and the water samples, was measured with high performance liquid chromatography (HPLC).

Table 1. Physicochemical characteristics of soil samples from the experimental plot.

Sample	S-30	S-60	S-90
Depth (cm)	30	60	90
Sand (%)	46	48	50
Silt (%)	36	36	34
Clay (%)	18	16	16
pH	6.25	8.88	9.39
C (%)	1.04	0.45	0.22
OM (%)	1.36	0.59	0.22

Table 2. Calculated concentration of adsorbed glyphosate in soil (Cs) from equilibrium concentration in the solution (Ce).

C (ng/ml)	S-30		S-60		S-90	
	Ce (ng/ml)	Cs (ng/g)	Ce (ng/ml)	Cs (ng/g)	Ce (ng/ml)	Cs (ng/g)
0	0	0	0	0	0	0
500	26	1185	32	1170	123	943
1000	58	2355	66	2335	279	1803
2000	107	4732	124	4690	577	3558
5000	319	11703	345	11637	1062	9845
10000	511	23722	813	22968	3670	15825

RESULTS AND DISCUSSION

The analytical results are presented in Figures 2, 3 and 4, which show the straight lines adjusted to the adsorption isotherms, the corresponding equation and the correlation coefficient (R^2).

The high values of R^2 (Table 3) show that the adsorption processes are well represented by the linear isotherms.

The fact that K_d decreases with depth suggests that there could be some relationship between the adsorption capacity and organic matter content, which also decreases with depth. However, K_{oc} (Table 3), which takes into account the organic fraction, is not in agreement with this alternative. This shows that in the case of the glyphosate, it would not be the organic matter content but the type of clays that is the predominant factor in the processes of adsorption and therefore the use of K_{oc} is pointless.

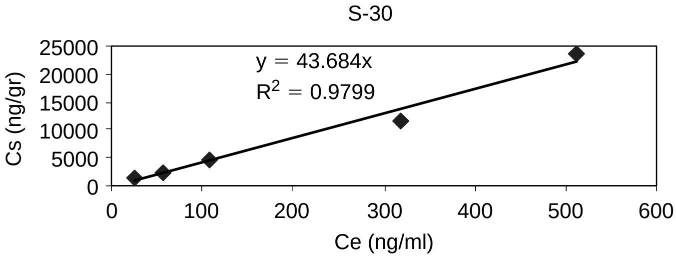

Figure 2. Adsorption isotherms at 30 cm depth.

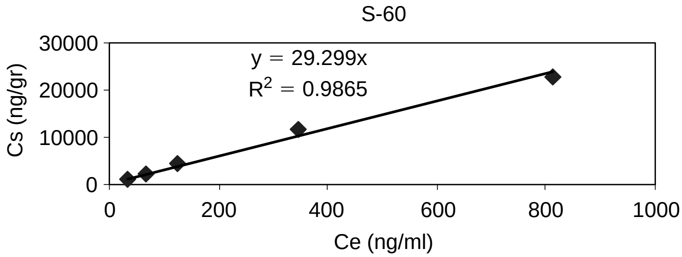

Figure 3. Adsorption isotherms at 60 cm depth.

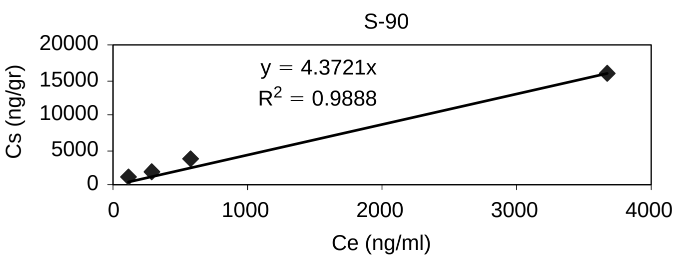

Figure 4. Adsorption isotherms at 90 cm depth.

Table 3. K_{oc} calculated from the percentage of soil organic carbon.

Sample	R^2	K_d	K_{oc}
S-30	0.9799	43.68	4183
S-60	0.9865	29.3	6484
S-90	0.9888	4.37	2023

Table 4. Freundlich isotherm adjustment.

Sample	R^2	N	ln K_F	K_F
S-30	0.9937	0.9841	3.8267	45.91
S-60	0.9970	0.9271	3.8964	49.22
S-90	0.9994	0.8344	2.8310	16.96

The results of K_{oc} obtained agree with those presented by Green & Karickhoff (1990). They claim that the use of K_{oc} is an approach that tends to unify the adsorption data for non-ionic hydrophobic compounds whose water solubility does not exceed 300 mg/l. The solubility of the glyphosate is higher than that and it behaves like a strongly ionic compound, which can even form complex compounds through its phosphoric group (Beltrán et al., 1995).

The calculated isotherms present good linearity. Nevertheless a certain degree of curvature is observed, mainly in the tests performed with samples S-60 and S-90, which suggests a better adjustment to the isotherm of Freundlich (K_F). In order to verify this alternative, a K_F adjustment was made (Table 4). As $n \rightarrow 1$, the isotherm linearity increases, making it possible therefore to admit that, at surface level S-30, the behavior is linear; in fact K_F and K_d are similar. At greater depths, S-60 and S-90 – there is less adjustment.

The glyphosate adsorption in this kind of soil is a process of great significance. A way to quantify the leaching potential of glyphosate and compare it with the other compounds is by means of the determination of the GUS index (Groundwater Ubiquity Score) (Gustafson, 1989), which is obtained from:

$$4\,\mathrm{GUS} = [\log t_{1/2}][-\log K_{oc}]$$

In this relationship $t_{1/2}$ is the mean compound life, which is 38 days for glyphosate (Morell & Candela, 1998). If we take a mean K_{oc} of 4000, the resulting GUS index is 0.6, which falls into the nonleachable category.

The velocity of a pesticide, without considering dispersion (Gerritse, 1993) can be calculated from the retardation factor (R) defined such that

$$R = \frac{v}{v_c} = 1 + \frac{\rho}{\theta} K_d$$

where v is the pore water velocity (cm/d), v_c is the pesticide velocity (cm/d), ρ is the bulk density (gr/cm^3) and è is the volumetric moisture content (cm^3/cm^3). Considering the conditions set during the tracer test, $v = 0.12$ cm/d, è $= 0.40\,cm^3/cm^3$ (Lexow & Bonorino, 1998) and $K_d = 43.7$, $R = 138$ for the first centimeters of the soil. Hence $v_c = 8.6 \times 10^{-4}$ cm/d, equivalent to the terms 'non-leachable' or 'strongly adsorbed'.

CONCLUSIONS

The glyphosate in the soil of the experimental plot underwent a significant process of adsorption that makes it, in terms of potential mobility, fall into the category of non-leachable.

The greatest adsorption of glyphosate took place at the most superficial soil layer and decreased with depth, due mainly to variations in the physical and chemical properties of the clay section and not so much to the effect of organic matter.

It seems useless to standardize the partition coefficient regarding the organic matter content for compounds which are highly soluble in water, possess strongly ionic behaviour and show that the adsorption mechanism does not take place exclusively on the organic component of the soil.

The non-leachable categorization would have to be adapted to the natural physical characteristics of each field because estimating the mobility of the glyphosate at sites with preferential paths is very difficult. The testing of adsorption in laboratories, although useful, has severe limitations.

The possibility that the front of pesticide concentration will move below the biologically active zone in a short interval of time does not agree with K_d as determined by this kind of test.

It would be necessary to conduct specific field studies in order to define the degree of interaction between the soil matrix and the pesticides in the surroundings of preferential flow ways. Those studies would help to establish the potential contamination of groundwater by such agents.

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CHAPTER 17

Hydrogeochemical simulation and experimental determination of Zn^{2+} transport in sediments at Mar del Plata, Argentina

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ABSTRACT: Infiltration experiments on undisturbed sediments in columns have been performed with the goal of calibrating a one-dimensional reactive transport model to be applied to Zn^{2+} mobility. The assays were carried out on sediments from a hilly region to the north of Mar del Plata City, Argentina. The sediments were carefully taken, driving the columns into the soil in order to minimize possible disturbance. The experimental chloride concentration curves were adjusted to the curves obtained using the code Phreeqc 2.0 for transport modelling and used to calibrate the model parameters. The values obtained were 0.01 m and 0.35 for dispersivity and porosity, respectively. A volume of $ZnCl_2$ solution equivalent to six times the pore volume of the column was passed through each column, measuring Zn^{2+} concentration after the arrival of each 0.5 pore volume. To achieve a satisfactory agreement between the observed and simulated Zn^{2+} curves it was necessary to consider the adsorption in the exchange surfaces and the equilibrium of the solid phase $Zn(OH)_2$ in the numerical model. The results showed that by the end of the experiment the main proportion of Zn^{2+} was adsorbed, remaining in the first 5 cm of the columns.

INTRODUCTION

Hydrogeochemical simulation provides a useful tool to predict water-composition variation and contaminant migration from the surface to the underground medium, as well as the mineralogy of a porous medium as a consequence of natural processes and system disturbances. Most hydrogeochemical surveys utilizing numeric simulation as a tool are oriented towards learning about the final destiny of certain contaminants in the underground porous medium. In order to predict soil and groundwater contamination risk in areas of high concentrations of contaminant elements, it is essential to know about the chemical behaviour of the elements, as well as about the mechanisms involved in their mobility. In surveys performed at the aquifer of Mar del Plata City, Argentina on contamination resulting from landfills (Bocanegra et al., 2001), zinc was the heavy metal detected in greatest concentrations in groundwater. This metal is less dependable on redox conditions than other metals such as iron and manganese. This fact, as well as its

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nutrient condition for vegetal species, makes it necessary to conduct surveys dealing with the mobility and behaviour of this heavy metal in the sediments of the area.

Several computerized numerical models performing hydrochemical simulation are based on general flow and transport equations. The flow equation for a saturated medium is as follows: (Appelo & Postma, 1993):

$$\partial C/\partial t = -v(\partial C/\partial X)_{\mathrm{t}} + DL(\partial^2 C/\partial \mathrm{X}^2)_{\mathrm{t}} - (\partial q/\partial t)_{\mathrm{x}} \quad (1)$$

where C is concentration, t time, x flow path, v pore flow velocity, DL the longitudinal dispersion coefficient and q the absorbed concentration. This equation has a first advective term, a second dispersive term and one of sink source, i.e. it contemplates the different ways in which a solute concentration moving in a pore medium could be affected.

Parameter and variable values included in the equation, among them dispersivity and flow velocity, are necessary to simulate solute transport in a saturated medium. Porosity is also a variable needed to obtain a precise calibration of the numerical model. Only some of these parameters can be identified and quantified in the field; thus in numerical simulation, reference values are generally used. For instance, in hydrogeochemical studies performed on the coastal area of Mar del Plata City, Argentina a dispersivity value of 200 m (Martínez & Bocanegra, 1998) was used to carry out simulations on a 1500 m long profile. In a survey conducted in China, with the aim of characterizing solute-transport properties in a sand and gravel aquifer, longitudinal and transversal dispersivity values obtained from field data (15 m long profile) were in the order of 1.72 and 0.0013 m, respectively (Yang et al., 2001).

In order to obtain better simulation outcomes, i.e. so that they represent the reality they intend to simulate as much as possible, it is convenient to obtain the specific parameters of the porous medium involved. A laboratory experiment widely used to estimate said parameters uses percolation tests of known concentrate solutions in columns containing a selected material and at flow velocities that can be regulated (Usunoff, 1999). Values estimated from laboratory tests are used in models as a first approximation; then adjustments, depending on the scale used, should be carried out (Usunoff, 1999). There is a relationship between the experiment scale and dispersivity values (Domenico & Schwartz, 1990). Depending on this relationship, at a macroscopic scale, values are generally two orders of magnitude above those of the experimental columns.

The aim of this work was to calibrate a solute-transport model in silty-sandy sediments in an area of Mar del Plata, Argentina, adjusting the parameters and variables required by means of experimental column tests and numerical simulation. Once the model was calibrated, the focus was on the study of Zn^{2+} transport by the sediments selected to predict this heavy-metal mobility, as well as the retention processes the involved sediments undergo.

MATERIALS AND METHODS

The sediments used in this work are part of an outcrop found in a soil profile located on Ruta National Route 226, 15 km away from Mar del Plata City (Figure 1). They are silt-sandy Holocene sediments of aeolic and fluvio-aeolic origin, widely distributed on the surface of the whole superficial northeasterly basin of the Tandilia range, corresponding to unit E3 according to Tricart (1973).

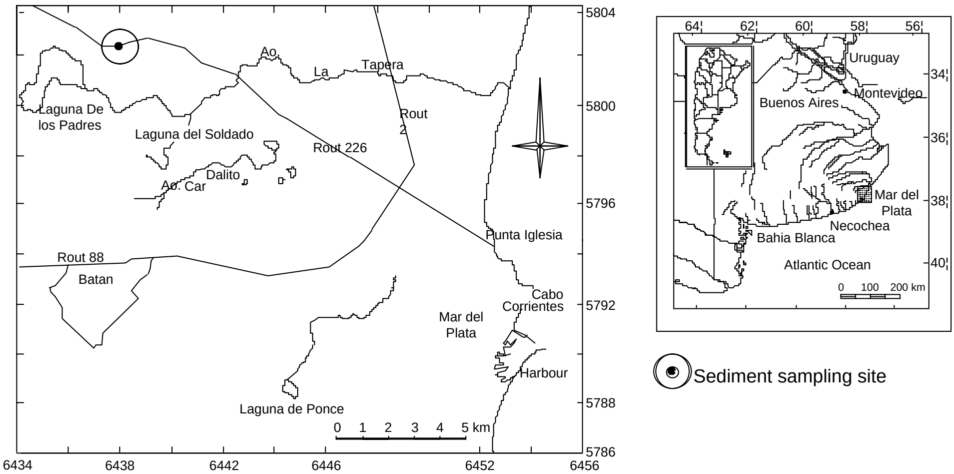


Figure 1. Geographic location of the sampling site.

Sediment samples were taken after shovelling 30 cm of surface soil (covered with vegetation). To perform the granulometric analysis, a sample was disintegrated and treated with hydrogen peroxide for as long as 48 hours, at first at a low temperature and later in a bath of boiling water at 40°C, until all the organic matter was eliminated. Texture was determined in the granulometric fraction above 62 μ (sands) using a dry sieving method, whereas pelitic material was analysed in the fraction below 62 μ (silts and clays) by pippet analysis. Statistical-parameter evaluation was performed following the methodology suggested by Folk and Ward (1957).

Total sediment porosity and cationic exchange capacity (CEC) were experimentally measured based on three samples, using the gravimetric method (Custodio & Llamas, 1976) and the direct displacement method (Rhoades, 1982), respectively. The CEC values obtained in meq · 100 g/soil were transformed into meq/l by means of the following equation (Appelo & Postma, 1993):

$$CEC(meq/l) = CEC(meq \cdot 100\,g/soil)10\rho/\varepsilon \tag{2}$$

where ρ is specific weight and ε porosity, expressed in moles/l.

The dispersivity value was experimentally estimated using a percolation test of NaCl solution of known concentration, in experimental columns containing silty-sandy sediments. To do so, three PVC columns, 23.5 cm long and 7 cm in diameter, were filled in the sampling site, driving the columns into the soil in a vertical position, so as to minimize disturbance. Once installed in the laboratory, a volume of distilled water equivalent to 1 pore volume of the column (250 ml) was passed through them and the leached liquid was collected and chemically analysed. The pore volume of the column was obtained following this equation:

$$Pore\ volume = column\ volume \cdot porosity \tag{3}$$

After saturating the columns with distilled water, a volume of (10 g/l) NaCl solution equivalent to five times the pore volume of the column was passed through. The flow obtained was vertical due to gravity and a dripping system was employed to regulate the flow in the column. Every 0.5 pore volume (125 ml), the concentration of Cl^- in the leached liquid was measured by means of the Mhor method. Dispersivity values were estimated, adjusting the experimental concentration curves for chloride to the curves obtained by simulation following the flow and transport model to be calibrated. Simulations with diverse dispersivity values were carried out until the best possible adjustment between the curves mentioned was obtained and, therefore, the most representative value of the parameter for the specified sediments.

Once the dispersivity value was adjusted to the specified sediments, a percolation test with a $ZnCl_2$ solution was conducted, so as to study the transport of this metal by said sediments. Columns were filled in a similar way to that previously mentioned and 6 pore volumes of $ZnCl_2$ (941.5 mg/l) solution were passed through them. The leached liquid was collected every 26 ml and every 0.5 pore volume (130 ml) the concentration of Zn^{2+} was determined by atomic spectrometry absorption. In both assays, electric conductivity was measured every 26 ml to determine the first arrival of the solution.

The computer program Phreeqc 2.0 (Parkhurst & Appelo, 1999) was used to perform the numerical simulation of one-dimensional solute transport, including diffusion and dispersion. This free-access program is widely used and boasts an important database for hydrogeochemical simulation. Moreover, the modelling code chosen allows the composition of the ionic fraction adsorbed in the exchange surfaces to be obtained, applying the concept of equivalent fractions. In this way, it is possible to analyse Zn^{2+} distribution between the solution and solid phase in each column cell (distance from the solution inlet) and in different simulation stages as well.

To simulate transport, Phreeqc 2.0 requires the characterization of the initial solution used to fill the column and of the solution as it passes through. To characterize the initial solution, the data obtained from the leached-liquid analysis when 1 pore volume of distilled water was passed through each column was collected. In all cases, apart from the chemical composition, the pH value and temperature had to be measured. Likewise, the program requires the column length to be defined and its theoretical division in a specified number of cells. Thus, the 23.5 cm long column was divided into ten cells, each 0.0235 m long. In addition, flow direction, limit conditions at the end of the column, dispersivity value and CEC value expressed in moles were reported. In Phreeqc 2.0 each step of transport is represented by the passage of the solution from one cell to the next and flow velocity is determined by the time, in seconds, that the solution takes to go through that passage. Each step of transport represents $1n^{-1}$ of the pore volume of the column, where n is the number of cells into which it is divided. Five and 6 pore volumes of NaCl and $ZnCl_2$ solution were passed through the columns in each assay and were expressed as 50 and 60 steps of transports respectively. Output data corresponded to the values obtained in the last cell of the column (cell 10) in each stage of transport.

In order to stimulate Zn^{2+} transport among the model-input data, the equilibrium with a $Zn(OH)_2$ solid phase was considered, as this is the most stable solid phase under oxic conditions of neutral or slightly alkaline pH (Stumm & Morgan, 1981; Deutsch, 1997). This was observed in Argiudol Vértico and Natracualf soils, typical of Buenos Aires Province, Argentina (Camilión et al., 1998), where Zn^{2+} appeared to be associated with oxides and linked to oxyhydroxides and oxides when this metal was increased in assays of experimental columns. The fraction associated with carbonates and organic complexes was less relevant.

RESULTS

The sediments studied are composed of 18.62 per cent fine sands, 65.29 per cent silt and 16.08 per cent clay, and the mean diameter of the particles is of 0.017 mm. The analysis of the statistical parameters demonstrated that the sediment had been very poorly selected.

The porosity value of the sediments was 0.24 and the CEC average value was 14.96 ± 0.92 meq · 100 g/soil. The CEC value converted according to Equation 2 was 1.38 ± 0.08 moles/l. Leached liquid collected from experimental columns when one pore volume of distilled water was passed through was analysed and was composed of 8.54 mg/l of Mg^{2+}, 28.5 mg/l of Ca^{2+} and 0.04 mg/l of Zn^{2+}. These values were used to characterize, in the numerical simulation, the solution initially contained in the column.

Experimental concentration curves for Cl^{-} followed the same pattern as those for the simulated curve. By adjusting these curves, a sediment dispersivity of 0.01 m was obtained (Figure 2a). The low dispersivity of the sediment was expressly shown by the

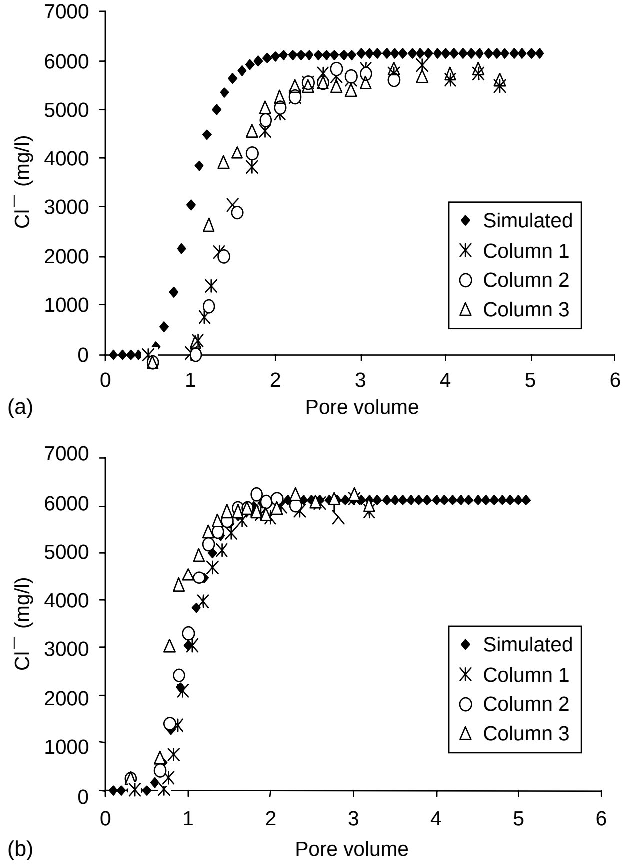


Figure 2. Simulated and experimental Cl^{-} concentration (mg/l) obtained in cell 10 in each column, in relation to the pore volume of a NaCl solution (10 g/l) passed through. The figure (a) considers a porosity value of 0.24 (pore volume 230.9 ml); and (b) a porosity value of 0.35 (pore volume 336.7 ml).

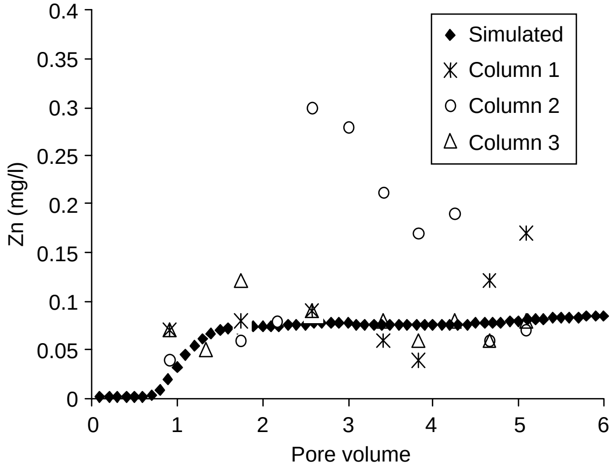


Figure 3. Simulated and experimental Cl^- concentration (mg/l) obtained in cell 10 in each column, in relation to the pore volume of a $ZnCl_2$ solution (941.5 mg/l) passed through (pore volume 302.5 ml).

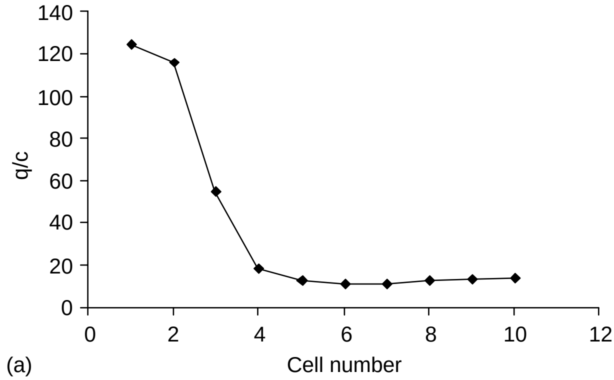
high gradient of the chloride curve. In spite of the fact that both curves were asymptotic in the inlet concentration value, at two pore solution volume outlet, the conservative element was observed in approximately 0.5 pore volume in the simulated curve and in 1 pore volume in the experimental curve (Figure 2a). Despite this dephasing, the parallelism between both curves indicates that the dispersivity value calibrated is appropriate. To improve the adjustment, the variables that could affect the outcome had to be considered and the change of porosity values then proved to have favourably modified the representation of the experimental curves for Cl^-.

The porosity value with which it was possible to obtain the best adjustment among the experimental and simulated data (Figure 2b) was 0.35.

Calibrated dispersivity and porosity values were considered for Zn^{2+} transport tests. Analytically obtained Zn^{2+} concentration values at column outlet were compared with numeric simulation outcomes using Phreeqc 2.0 (Figure 3). The Zn^{2+} values measured in the experiment were similar to those obtained by simulation. This shows that dispersivity and porosity values estimated by Cl^- transport assays are valid for the objectives suggested. The best adjustment of the solution initially contained in the column was further achieved by using a Zn^{2+} concentration value in the numerical simulation of 0.015 mg/l, instead of the measured value of 0.04 mg/l, and by considering the equilibrium with a solid phase of $Zn(OH)_2$.

Figure 4 represents the distribution relationship of adsorbed Zn^{2+} concentration (q) and Zn^{2+} concentration in solution (c) at 10 and 60 steps of transport, i.e. after passing through column 1 and 6 pore volumes of $ZnCl_2$ solution respectively.

Zn^{2+} was mainly adsorbed in a Zn-X_2 way (where X is an exchange surface) in a proportion ranging from 120 to ten times the solution concentration. Figure 4 also shows the displacement of the highest adsorption sector towards the column outlet (from cell 1 to 3–4) between the passage of the first and sixth pore volume.



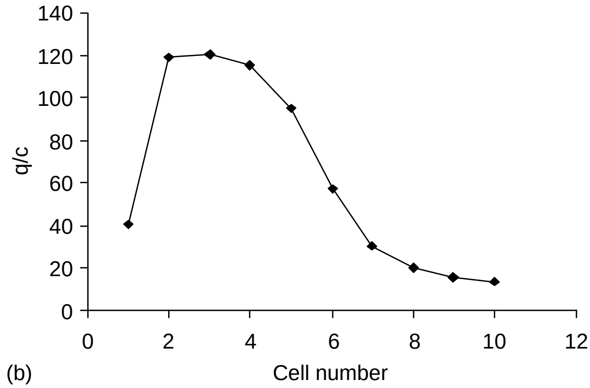


Figure 4. Relationship between the adsorbed Zn^{2+} concentration (q) and Zn^{2+} concentration in solution (c) obtained by simulation in each of the ten cells of the experimental columns at (a) step 10 of transport and (b) step 60 of transport.

DISCUSSION

The dispersivity value for silt-sandy sediments in the area of Mar del Plata was obtained from the calibration of the numerical model or following the experimental data for Cl^- transport. The value of dispersivity, which allowed a better adjustment of the experimental to the simulated for Cl^-, was 0.01 m. This value is between the typical range of dispersivity values used for experimental breakthrough columns tests, quoted by Doménico and Shwartz (1990). Since calibration was performed manually by trying different dispersivity values, some variations in this parameter could be expected if an automatic calibration were done instead. Not withstanding this, the total parallelism among the curves shows that the 0.01 m value can be considered representative. The use of other numerical models for transport simulation to verify the validity of the theoretical curve obtained should not be discarded.

The experimentally measured porosity value had to be modified to obtain a better adjustment of the experimental concentration curves for Cl^- to the simulated ones. The value with which the best adjustment was obtained was 0.35, a value above that obtained experimentally (0.24). This difference has two probable causes: on the one hand, it is possible that the fact that the columns were driven into the soil (in the field) slightly modified the medium porosity due to spatial generation around the edges. On the other hand, experimental determination of porosity was carried out on relatively small sediment fragments (1 cm^3), while in the columns there were other spaces only appreciated at a larger scale, such as macropores mainly generated by the presence of roots in the natural landscape. Thus, it is likely that the porosity value in the columns may be even greater. Once calibrated, both parameters (dispersivity and porosity) and the achieved adjustment for conservative transport (Figure 2b) turned out to be satisfactory.

The concentration values of Zn^{2+} obtained in the columns outlet when 6 pore volumes of $ZnCl_2$ solution were passed through were within the same order of magnitude as the simulated values. The dispersivity and porosity values previously obtained seem to be appropriate for the sediments under analysis, even though certain aspects of the Zn^{2+} geochemistry in this medium should be studied further to explain the differences with the numerical model. The greatest differences found between Zn^{2+} concentration values in columns outlet 2 and the simulated values were attributed to an error in the experimental conduction.

Zn^{2+} transport simulation results using Phreeqc 2.0 showed that the main processes controlling Zn^{2+} mobility in the medium are: (i) adsorption in exchange surfaces and (ii) solid phase equilibrium with Zn hydroxide ($Zn(OH)_2$). The first of these processes is determinant. The displacement of the greatest adsorption sector towards the column outlet (from cell 1 to 3–4) between the passage of the first and sixth pore volume of $ZnCl_2$ solution (Figures 4a and 4b) is a typical effect of retardation caused by adsorption of a reactive element in the medium, with a distribution coefficient accounting for a marked tendency to be adsorbed.

The analytical determinations performed on the resulting solution after passing 1 pore volume of distilled water through the columns, whose results yielded concentration values of Zn^{2+} of 0.04 mg/l, show that this element occupies adsorption surfaces in the uncontaminated sediment and that it is released in a very low ionic force solution. The initial concentration of Zn^{2+} used in the simulations to characterize the solution initially contained in the column must have been lower (0.015 mg/l) than that measured (0.04 mg/l) to obtain a good adjustment of the experimental curves to the simulated ones. It could be assumed that the 0.015 mg/l is the concentration value resulting once 1 pore volume of distilled water is passed through, as the concentration of 0.04 mg/l initially measured would decrease progressively with distilled water passage, according to the expected behaviour for any cation in a typical elution curve (Appelo et al., 1993). The dilution effect was confirmed by batch-like tests performed on the sediment. Different initial zinc concentrations showed a more than half reduction of said concentration in solution at 24 and 72 hours of contact. Currently, an elution assay in experimental columns is being performed to confirm these outcomes.

CONCLUSIONS

Knowing about Zn^{2+} mobility and retention processes of silty-sand sediments in Mar del Plata City, Argentina is important to assess the risk of soil contamination by heavy metals, as well as the potential contamination of groundwater. Because of this, the experiments conducted with dynamic assays in laboratory, which enabled the parameter values necessary for contaminant transport simulation to be obtained, are an interesting first approach to study these problems in sediments, for which there is no previous information.

The curve obtained by simulation (using Phreeqc 2.0), following a dispersivity value of 0.01 m and a porosity value of 0.35, is well adjusted regarding the experimental concentration curves for the conservative element. It has been proved that these values are adequate for reactive transport simulation; however, for elements such as Zn^{2+}, it is necessary to identify the processes controlling their mobility. It was concluded that for the silty-sandy sediments studied, adsorption was the main process and that, to a lesser extent, the equilibrium kept with the Zn^{2+} hydroxide solid phase exercises control over the concentration of the solution.

Zn^{2+} concentration in the solid phase is between 120 and 10 times greater if compared to the liquid phase of the columns analysed. Almost all the Zn^{2+} introduced in the inlet solution, after 6 pore volumes, remains in the first 5 cm of the column. Taking into account that the sediment employed is silty-sandy and not very clayey, it is possible that other much more clayey sediments in the area constitute a high retention barrier for heavy metal.

It is now necessary to study this subject in depth, applying other numerical models and conducting new laboratory and field tests to predict Zn^{2+} mobility in pampean sediments, as well as to perform an automatic calibration of the parameters the model requires.

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CHAPTER 18

Hydrochemical characterization of ground and surface waters in 'the Cotos' area, Doñana National Park, southwestern Spain

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ABSTRACT: The Doñana National Park holds one of the most important wetland complexes in southern Europe. It is located in the right margins of the River Guadalquivir, in the south west of Spain. Groundwater plays an important role in the functionality of the ecosystems. The aquifer system consists mainly of unconsolidated Plio-Quaternary materials covering an area of 3400 km^2. In the studied sector there are two main lithostratigraphic detrital units: the Eolian Unit, which is a homogeneous layer of dominantly siliceous sands containing low mineralized groundwater of the Na-Cl type, and the Deltaic Unit, which is a discontinuous level of siliceous sands, silts and gravels, where some carbonate remains locally, containing groundwater of the Na-Ca-HCO_3-Cl type. In the Eolian unit and around some lagoons there are more concentrated and even brackish phreatic groundwaters due to evaporation by phreatophytes or affected by mixing with evaporated lagoon water. Isotopically, groundwater has the local rainfall signature (even that affected by phreatic evapoconcentration) while lagoon waters show isotopic enrichment due to evaporation.

INTRODUCTION

The Doñana National Park includes one of the most important wetland complexes in southern Europe. There is a large diversity of ecosystems in the area and their functioning is closely related to the hydrogeology of the area (Manzano, 2000; Manzano et al., 2002). This park is placed in the natural region of Doñana, in southwest Spain, in the confluence of Huelva, Sevilla and Cádiz provinces, partly occupying the palaeo-estuary and delta of the River Guadalquivir (Figure 1). Three morphogenetic systems are included here: deltaic, littoral and eolian (Zazo, 1980; Siljeström et al., 1994; Rodríguez-Ramírez et al., 1996). Two geomorphological units have been identified in the eolian system: stabilized dunes and mobile dunes (Borja & Díaz del Olmo, 1994; CMA, 1998).

The study area, locally called 'the Cotos' (the hunting reserves), corresponds to the western section of the Doñana National Park and also includes some areas in the park (Figure 1). Its boundaries are the coastline to the south west, the road connecting the village of El Rocío to the tourist resort of Matalascañas to the northwest, and the ecotone (contact between the eolic sands and the clays of the marsh) to the east. The presence of an active dune string parallel to the coast, coexisting with stabilized dunes, is highly remarkable, not to mention the existence of several peridunar, permanent and seasonal, phreatic lagoons in the topographic depressions.

Several hydrogeological studies have been carried out in the Doñana area from the 1970s to the present day. Rodríguez-Arévalo (1984), Vela (1984) and Tenajas (1984) studied some aspects of the hydrogeology of the Doñana National Park. Baonza et al. (1984) carried out an isotopic study of the aquifer, subsequently completed by Poncela et al. (1992) and Manzano et al. (1991, 2001, 2003). Vela et al. (1987) and Sacks et al. (1992) carried out detailed studies on some lakes of the Doñana National Park. The studies of Custodio et al. (1992), Custodio (1992, 1994) and Salvany and Custodio (1995) defined the geological and hydrogeological framework of the aquifer. Iglesias (1999) and Trick (1998) carried out two complementary studies defining other hydrogeological aspects of the zone.

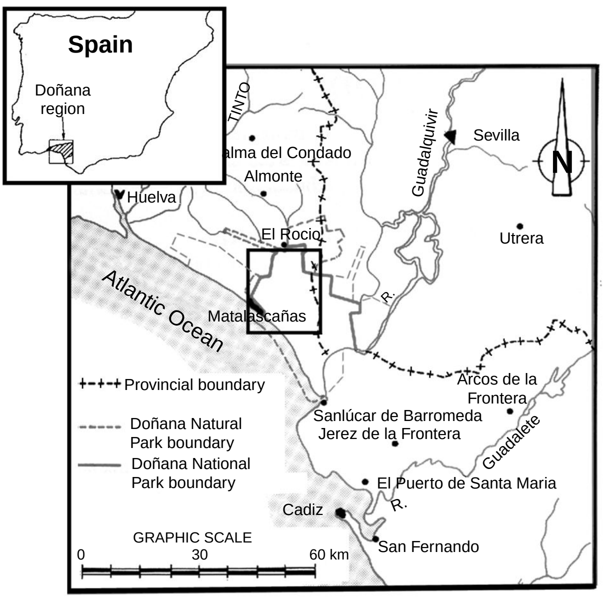

Figure 1. Geographical situation of the study area.

GOALS

The goals of the study presently under way are: i) to characterize the isotopic (^{18}O and ^{2}H) and chemical (major elements) composition of the groundwater and the lagoon water in the study area; and ii) to determine the lagoon's hydrodynamic behaviour and to characterize the hydrochemical processes in the aquifer. To complete this interpretation, the values of some physicochemical parameters measured in the field have been taken into account, such as pH, electrical conductivity (EC) and water temperature (T). Minor ions and trace components, as well as other environmental isotopes, are being studied in parallel but results are not included in this paper.

MATERIALS AND METHODS

Most of the groundwater samples are from exploratory boreholes and from small diameter (100 mm) piezometers of different depths (between 10 and 150 m), usually with short-screened boreholes in the bottom. Sixty-two boreholes in the Eolian Unit were sampled and 22 from the Deltaic Unit (these units are defined later on). The samples were taken with a 0.5 and 1L Blassi® sampler in the screened interval. The phreatic samples (depth < 2 m) were sampled at 48 points as part of the doctoral thesis of the first and last authors, as part of the same research project. The samples of lagoon water were taken directly, always in the same places. To describe groundwater, phreatic water and lagoon water, major ions content and isotopic data were used. The water samples were taken from these boreholes in the period 1990–2001. The phreatic and lagoon water samples were from the period 1998–2001.

The certified chemical laboratory of the IGME (Geological Survey of Spain) in Madrid carried out the analysis of the major elements and other solutes using standard and homologated methods. The stable isotopes, ^{18}O and ^{2}H, were analysed at the CEDEX laboratories in Madrid, Spain and at the GGA (Institut für Geowissenschaftliche Gemeinschaftsaufgaben, Hannover, Germany).

The physicochemical parameters were measured in the field: pH and EC (electrical conductivity) with a Hanna® pH-meter and a Hanna® conductivimeter, temperature with a mercury thermometer and alkalinity with a Hach-titration® kit.

HYDROGEOLOGICAL CONTEXT

The study zone is part of the aquifer system of Almonte-Marismas, or Doñana, formed by Plio-Quaternary sandy materials, of a mainly siliceous nature, occupying around 3400 km^2. The lower boundary consists on Miocene and Pliocene marls (Figure 2).

In the study zone includes the Eolian Unit, the Deltaic Unit and the Marsh Unit, defined by Salvany & Custodio (1995) (Figure 2).

The Eolian Unit is a relatively homogeneous, sandy package, with thin and occasional clayey and silty intercalations. The thickness of the unit is around 150 m at the coastline, decreasing to a few metres inland. It consists mainly of amorphous silica and quartz, with scarce K-feldespar (microcline) and Na-feldespar (albite), and some clayey materials (illite, kaolinite, chlorite and clinochlore). Traces of muscovite, andalusite and tourmaline have also been identified; carbonate minerals are absent (Iglesias, 1999).

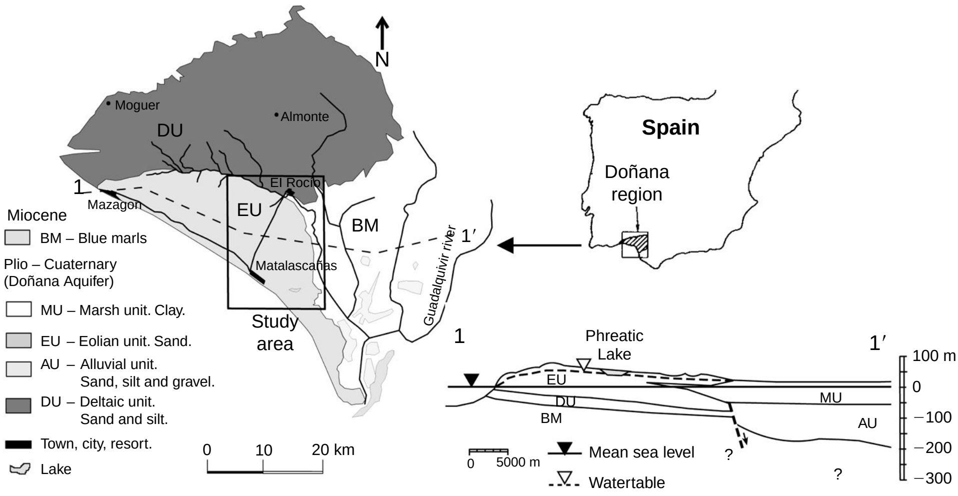

Figure 2. Geological map of the aquifer system in Doñana and cross-section along the study area.

The Deltaic Unit is formed by a discontinuous layer of siliceous sands, silts and gravels, with a clay matrix. It underlies the Eolian Unit and grades into the Pliocene marls underneath. Its thickness ranges from 6–70 m in the El Rocío area (Figure 2) down to a few tens of metres in some areas of the coastline. The dominant minerals are also amorphous silica and quartz, although muscovite, Na-feldespar and kaolinite are also found in some areas. The main mineralogical difference with respect to the Eolian Unit is the presence of biogenetic calcite in a bioclastic layer found to the north and north east of the studied area (Salvany & Custodio, 1995; Iglesias, 1999).

The Marsh Unit is formed by clays and silts with occasional layers of gravels, sands, peat and bioclasts, and has a thickness between a few metres and 50–70 m. On its western side this unit gradually interleaves with the Eolian Unit (Salvany & Custodio, 1995), shaping the ecotone.

Local rainwater is the aquifer recharge water source. Mean annual precipitation ranges between 500 and 600 mm. The main groundwater discharge areas are the coastal zone, the La Rocina creek and the contact area between the Eolian and the Marsh Units (the ecotone). On a local scale there is also groundwater discharge to the peridunar phreatic lagoons, where most of it evaporates.

The deep layer (Deltaic Unit) has a higher transmissivity and hydraulic diffusivity than the overlaying sands (Eolian Unit), which contains the water table (Salvany & Custodio, 1995; Custodio & Palancar, 1995; Trick & Custodio, 2003). In this area the aquifer is unconfined, while to the south east the permeable sediments are confined under the Marsh Unit. The interleaving of eolian and marsh materials frequently occurs near the ecotone, causing the aquifer to behave as a multilayer system locally.

RESULTS AND DISCUSSION

In general, groundwater from the study zone shows a low level of mineralization. The median of the EC is 185 μS/cm. This low content of dissolved solids is related with the

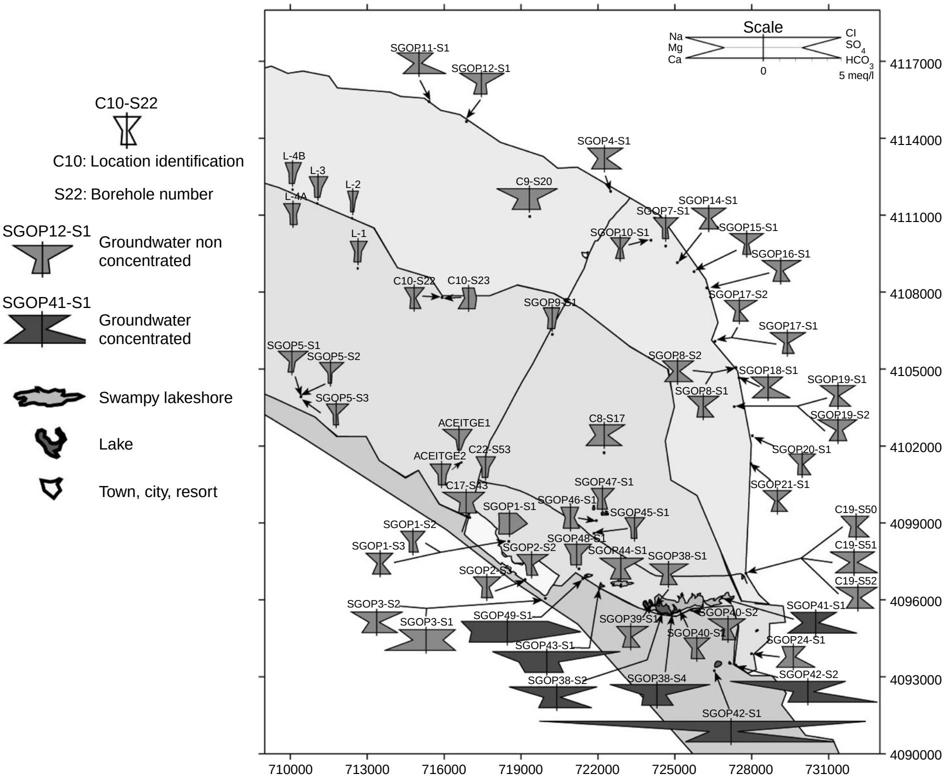

Figure 3. Modified Stiff diagrams for the groundwater of Na-Cl type. Two groups of groundwater are differentiated: non concentrated groundwater and concentrated groundwater due to: i) mixing with evaporated lagoon waters; ii) evapoconcentration of phreatic water by phreatophytes; and iii) uptake of precipitated salts in peripheral areas of the lagoons.

small reactivity shown by the silica sands forming the aquifer and the relative high recharge in the sandy area. Some high EC values (up to 1700 μS/cm) are found in some areas. They are mostly the result of evapoconcentration around the lagoons, which are densely vegetated and more retentive of rainwater, and may contain remnants of evaporated lagoon water from high stages or precipitated salts. Groundwater pH shows a median (and mean) value of 6.5 in the carbonate-free sands, but it is almost one unit higher locally when groundwater is affected by $CaCO_3$ dissolution.

Major elements

This study intended to establish the characteristic chemical composition of the area according to the major elements, differentiating the two above-mentioned geological units – the Eolian and Deltaic Units. In the study area the Deltaic Unit is discontinuous. Two groundwater groups were identified: one group belongs to the Na-Cl type and the other to the Na-Ca-HCO_3-Cl type.

Figure 3 shows the spatial distribution of the chemical composition belonging to the sampled boreholes in the Eolian Unit. Most of these boreholes have water of the Na-Cl type due to the airborne marine influence in local rainwater. The modified Stiff diagrams define an area to the north west where groundwater has a low mineralization with the local rainwater signature, with small modifications. The Na-Cl component in the coastal zone probably increases due to the proximity to the coast, where the influence of airborne marine salts is intensified. This same Na-Cl component is also observed in the ecotone area in contact with the marsh, but at the same time the Ca and HCO_3 concentrations increase, possibly associated with either the mixing with discharge of carbonate-rich deep groundwater or the presence of carbonated biogenetic remains in the eolian sands.

The chemical composition of some of the groundwater around the peridunar lagoons differs from the rest of the groundwater described above. These are waters of different types, mainly of the Na-Cl type, although there are also waters of the Na-HCO_3-Cl, Ca-Na-HCO_3-Cl, Na-Cl-HCO_3 types, and at one point of the Na-SO_4-Cl type (SGOP49-S1 in Figure 3). These last waters are more saline, with an EC range of 400–1700 μS/cm. Some of them represent a mixture of lagoon waters with the surrounding groundwater, but it seems that this mixture only takes place at some points and in the first metres of the soil, since this phenomenon is not reflected at a greater depth or in the sampled boreholes of what was assumed to be the underground lake discharge area. This was verified by the phreatic water samples (depth < 2 m), taken in the surroundings of the lakes (Delgado et al., 2001a, 2001b; Lozano et al., 2001, 2002). Other waters may be explained by evapoconcentration of phreatic water by phreatophytes or when rain dissolves salts precipitated from evaporating water at dried-up peripheral areas of the lagoon, before accumulating in the **inundated** lagoon area.

The NO_3 ion was used as an indicator of anthropic activity in the aquifer. Higher concentrations of this ion were observed in the shallow groundwater than in the deep groundwater. This is mainly related to recent intense agricultural activity in some fringe areas of the Doñana National Park. Inside the protected area high NO_3 content is restricted to areas around human settlements (a number of houses inside the park are still inhabited).

Figure 4 shows groundwater in boreholes in the Deltaic Unit. Groundwater here is mainly of the Na-Ca-HCO_3-Cl type due to biogenetic calcite dissolution, which is found locally deep in the aquifer (Manzano et al., 1991; Iglesias et al., 1996; Iglesias, 1999). This groundwater is characterized by a slightly higher mineralization than the Na-Cl water type. Deep groundwater from the ecotone strip (samples to the north east in Figure 4) have the higher content of Ca and HCO_3. Points with Na-Cl waters, as those of El Acebuche (Aceb 2, 3 and 4), reflected a mixture of groundwater from the Eolian and Deltaic Units. The coastal samples were also of the Na-Cl type, although they also had a higher content of Ca-HCO_3 than the groundwater of the Na-Cl type.

During the study period (1998–2001), the EC of the lagoon waters ranged between 0.2 and 28 mS/cm (Figure 5). The observed increase in EC during this period was assumed to be due to the low precipitation, which caused a significant reduction of the swampy surface in many of the lagoons and even the drying up of some of them. There was only one dilution event during the wet period. Figure 5 shows the evolution of EC in three of the lagoons studied – the Charco del Toro (now a temporary lagoon) and the Dulce and Santa Olalla (permanent lagoons) (Figure 6). These three lagoons were chosen because the information on them was more complete than for others, compiled on a monthly frequency by the last author of this paper. The waters from the Dulce and Santa Olalla

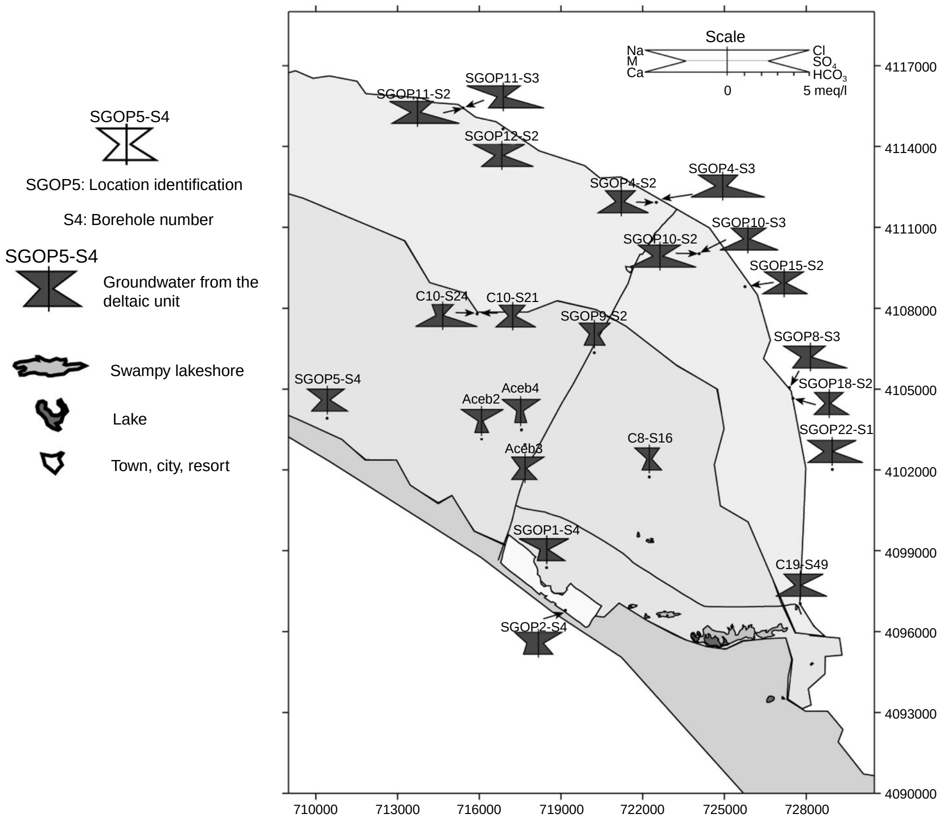

Figure 4. Modified Stiff diagrams for the groundwater of the Na-Ca-HCO_3-Cl, Na-Cl and Ca-HCO_3 types.

lagoons showed an increasing EC from March 1998 to October 1999. Santa Olalla lagoon had higher EC values, except in August and September 1999, when the Dulce lagoon almost dried out. The Charco del Toro lagoon was practically dry during the whole study period. It underwent a continuous rise of EC when it had water from March to October 1998.

The chemical composition in the Dulce and Santa Olalla lagoons belongs to the Na-Cl type, although they showed different concentrations.

The evolution of the chemical composition of these lagoons from March 1999 to May 2000 is shown in Figure 7. They showed more diluted water in March 1999 and reached the highest concentration in August 1999. A temporary rise of the rSO_4/rCl ($r = meq/l$) ratio was observed for October 1999 in both lagoons, which was simultaneous to the dilution of the lagoon due to the early autumn rainfall (see Figure 5). In May 2000 the lagoons again showed the same diluted chemical composition that occurred in March 1999. This evolution shows how the concentration of the major ions is controlled by rainfall and groundwater contributions and the evaporation-dilution cycles, except for the sulphate, which is affected by processes involving the oxidation of reduced sulphur deposited in the lagoon-associated sediments, mentioned later.

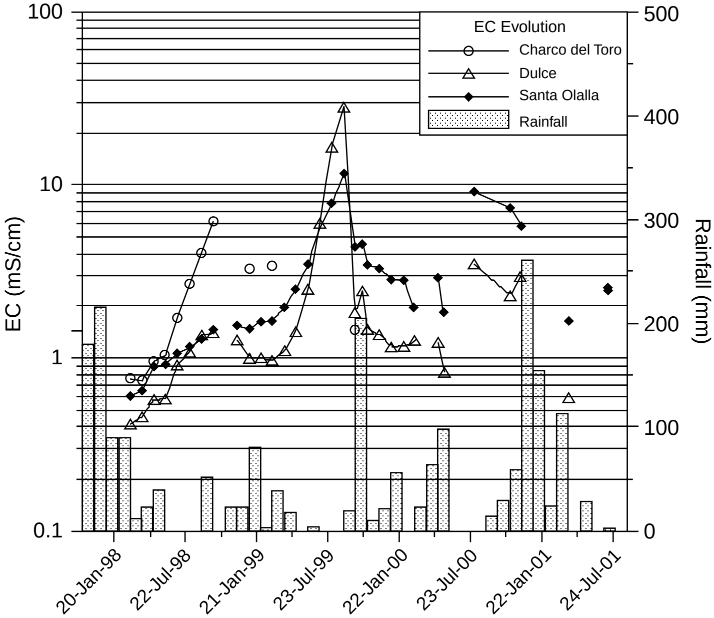

Figure 5. Evolution of electrical conductivity (EC) in three of the studied lagoons – Charco del Toro (temporary), Dulce and Santa Olalla (permanent) – and of the monthly precipitation in the study period.

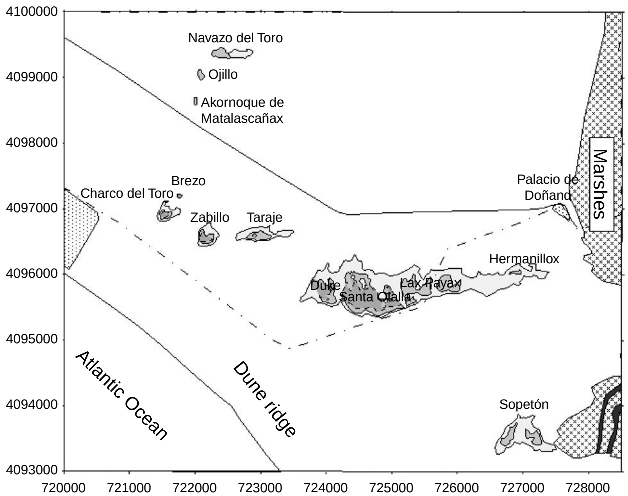

Figure 6. Postitions of the peridunar lagoons.

The water composition of the Charco del Toro lagoon (dry during nine months in 1999 and during the whole of 2000) evolved from Na-Cl in March 1998 to $Na-SO_4-Cl$ in March 1999 (Figure 7). The change in the composition took place in October 1998, although in this case the water EC was still high due to the lack of significant dilution,

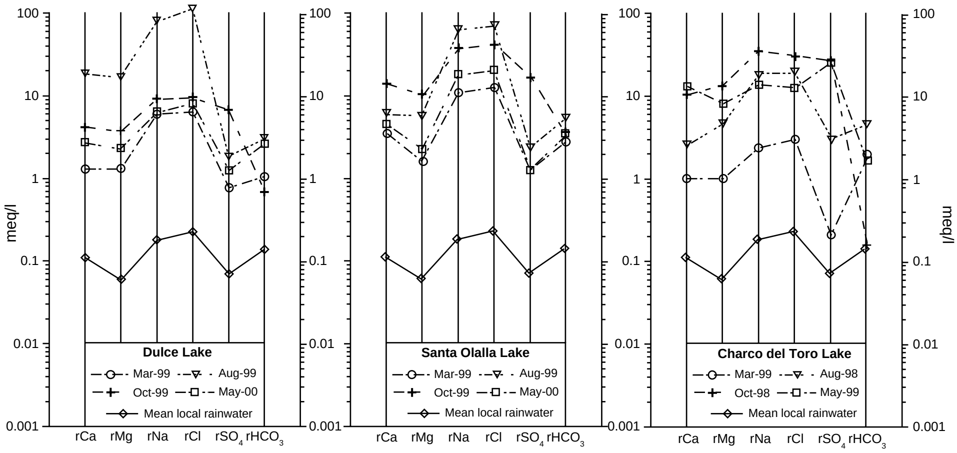

Figure 7. Logarithmic diagrams showing the evolution of the chemical composition of the Dulce, Santa Olalla and Charco del Toro lagoons water during the study period.

caused by the scarce rainfall registered during these months compared with 1999 (Figure 5). Na-SO_4-Cl waters were also present in the flood periods in 1999 (Figure 5). This evolution was also observed in the Zahillo lagoon (south east of Charco del Toro), which was periodically dry (see Figure 6).

The possible origin of the sulphate enrichment found in some surface waters can be related to the combination of two processes, currently being studied.

1. Oxidation to sulphate of reduced sulphur present in the lagoon sediments. Fluorescence and R-X difraction studies show that lagoon-bottom sediments, rich in organic matter, contain a significant concentration of non-mineral, reduced S, which would oxidize and be dissolved as sulphate by the early autumn rainfall later on. This effect seems more intense in lagoon areas affected by water-table drawdown, where organic-rich deposits are being oxidized. The existence of old dried-up bottom layers, rich in organic matter, probably plays a major role.
2. Local precipitation of gypsum in the swampy lake shore during the drying up period and subsequent dissolution in the next wet period.

Studies are currently going on to confirm the existence of gypsum in the swampy areas of some lakes, and to determine the origin and evolution of S in the sediments. It seems that recent groundwater has a higher SO_4/Cl ratio than recharge produced a few decades ago due to increased airborne industrial pollution, mostly from the Huelva industrial area, where sulphide minerals from the Iberian Pyrite Belt are burned. But this is still to be checked in detail.

In both situations the lagoons' sediments would act more or less as a temporary trap of S during the wet periods. This means that, while other elements such as Na and Cl may be exported during some periods by groundwater flow, S is retained either in the organic lagoon-bottom sediments or as gypsum in the flood areas, ready for its subsequent oxidation and re-dissolution. This is supported by the observation of low sulphate content in the lagoon waters in summer periods such as August 1999. The microbiological reduction of S may be favoured by the elevated summer temperatures when the lagoon water reaches 30°C (Sacks et al., 1992), as measured during the summers of 1998 and 1999 in the Dulce and Santa Olalla lagoons.

Stable isotopes ^{18}O and ^{2}H

Previous research (Baonza et al., 1984; Vela et al., 1987; Manzano et al., 1991; Iglesias, 1999; Manzano et al., 2001) found that groundwater in the unconfined area had the isotopic signature of local rainfall: $\delta^{18}O = -4.7$ to -5.0‰ SMOW (standard mean ocean water) and $\delta^{2}H = -28$ to -30‰ SMOW.

Most of the $\delta^{18}O$ and $\delta^{2}H$ groundwater data in this study points to $\delta^{18}O = -5.0 \pm 0.3$‰ SMOW and $\delta^{2}H = -30 \pm 3$‰ SMOW (Figure 8). The **deuterium** excesses is close to +10‰ and indicates a rainwater recharge without significant evaporation.

The surface water samples from the peridunar lagoons in Figure 6 plot along an evaporation line with slope 4.4 (black discontinuous line in Figure 8), which is characteristic of the evaporation of surface water. Some groundwater samples, both phreatic (depth <2 m) and deeper (<20 m), also lie on this evaporation line. This may suggest that they were sampled on a flow path downflow from the lagoons, possibly representing the mixing of surface waters and local groundwater in different proportions.

The $\delta^{18}O$ values were represented versus the Cl and SO_4 concentrations, making a distinction between the phreatic (depth <2 m) and shallow groundwater (<20 m), and the surface waters. The Cl-$\delta^{18}O$ relationship (Figure 9) shows that most of the groundwater samples increase salinity without isotopic enrichment (indicated in Figure 9 by the arrow). This is due to rainfall concentration by evapotranspiration of plants or to the dissolution of previously precipitated salts in peripheral areas of the lagoons by

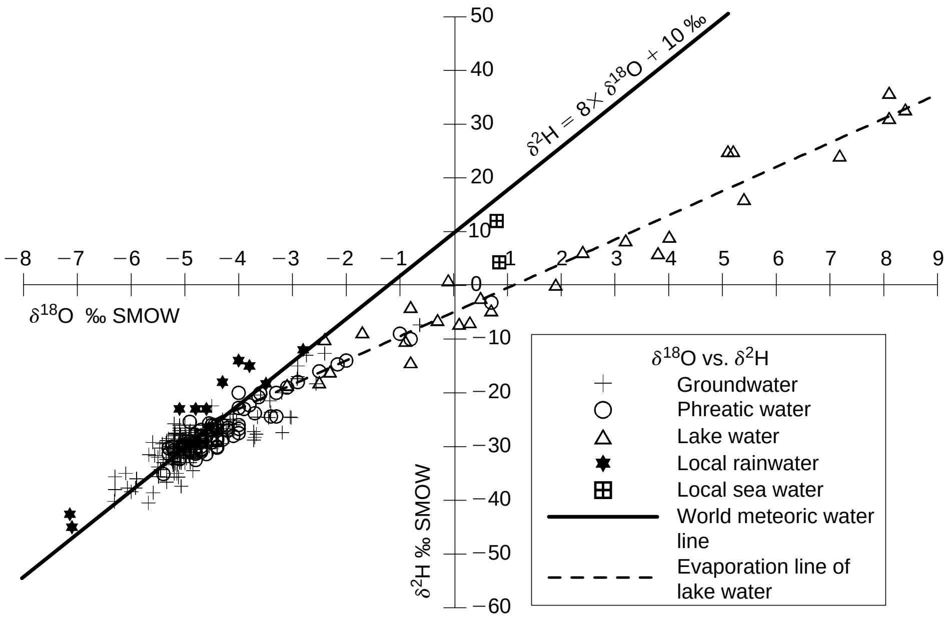

Figure 8. Diagram $\delta^{18}O$ versus $\delta^{2}H$ of groundwater, phreatic waters and surface waters.

isotopically unfractionated rainwater. This phenomenon was also observed in most of the phreatic water samples.

All lagoon waters showed the effect of isotopic fractionation by evaporation in different degrees, but starting from a variably concentrated water by phreatic evapoconcentration or dissolution of precipitated salts. This results in a wide range of Cl and $\delta^{18}O$ content (points inside the dotted line in Figure 9).

Some phreatic water samples underwent isotopic enrichment and formed part of the group of evaporated lagoon waters (Figure 9). These phreatic waters were sampled in the surroundings of the Santa Olalla lagoon, in sites suspected as possible underground outflow paths of lagoon water. Therefore, they represented the mixture of groundwater and evaporated surface waters from the lagoons (Lozano et al., 2001, 2002).

As for the SO_4-$\delta^{18}O$ relationship, the same process was observed. Figure 10 shows that most of groundwater and phreatic water had a wide range of SO_4 concentration (1–1000 mg/l), while keeping a similar isotopic composition. The samples inside the dotted line (Figure 10) refer to surface waters with different evaporation/dilution degrees. It was observed that the isotopically heaviest waters and the lowest SO_4 content (30–50 mg/l) in lagoons waters coincided with low flood levels, while the isotopically lightest waters and a wide SO_4 content (10–1000 mg/l) matched the highest flood levels (Lozano et al., 2001, 2002). These figures agree with the chemical evolution of some lagoons, which shows an important increase in the SO_4 concentration after the first autumn rainfall. Figure 11 displays the evolution of the SO_4 and $\delta^{18}O$ concentrations in three of the studied lagoons (Dulce, Santa Olalla & Las Pajas; see Figure 6) and the rainfall. It shows how in the rainy periods (October 1999 and December 2000) there was an increase in the sulphate content (dotted line in Figure 11) and the water became isotopically lighter (continuous line in Figure 11); however, in dry periods the isotopic composition was isotopically heavier and the SO_4 content low.

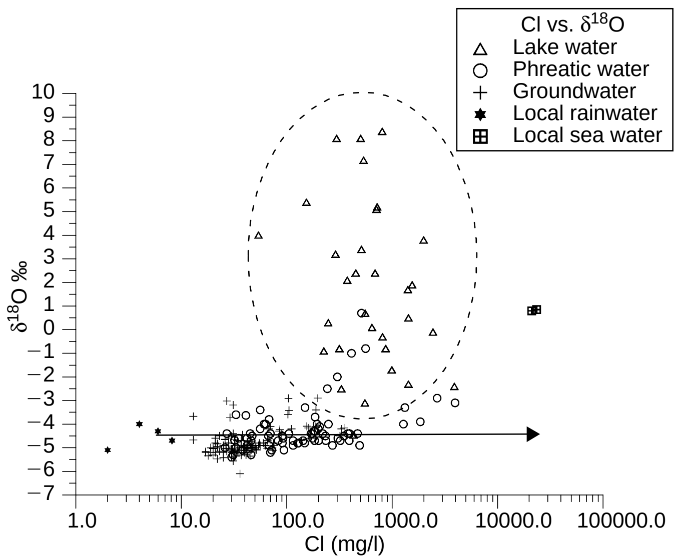

Figure 9. Cl versus $\delta^{18}O$ in groundwater, phreatic waters and surface waters of the study area.

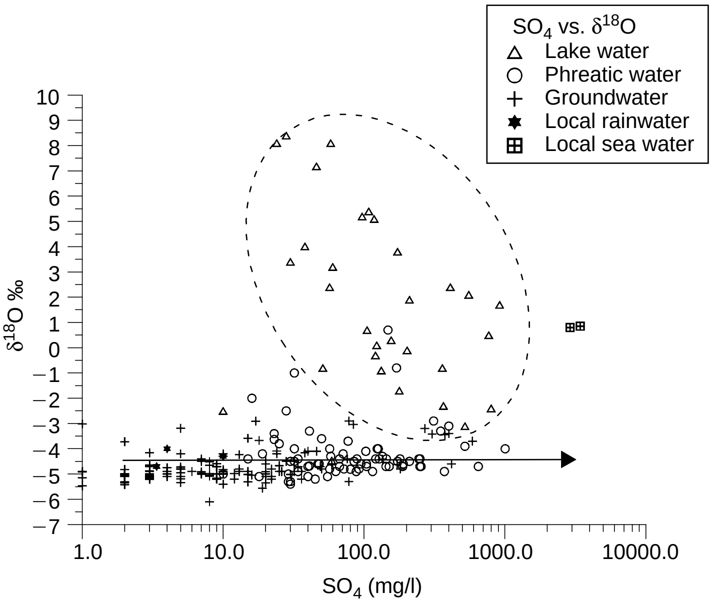

Figure 10. SO_4 versus $\delta^{18}O$ in groundwater, phreatic waters and surface waters in the study area.

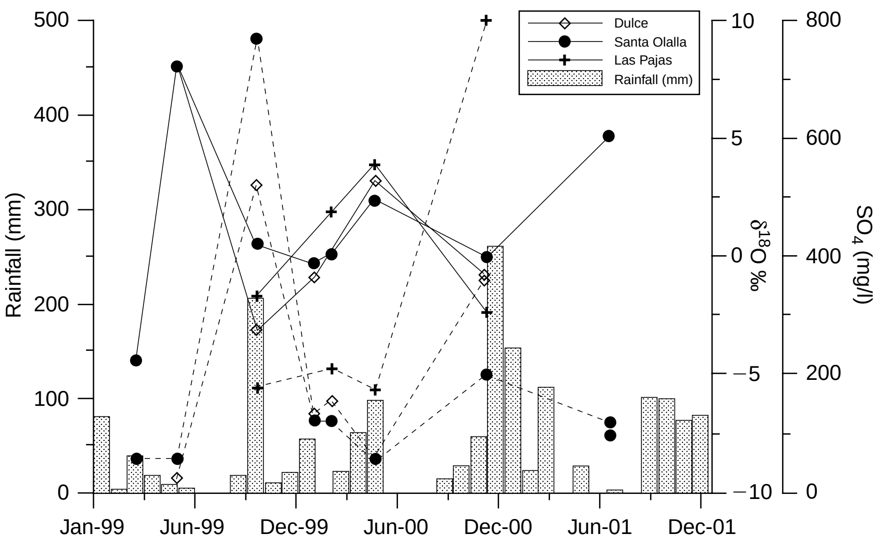

Figure 11. Evolution of the SO_4 content and $\delta^{18}O$ in surface waters from the Dulce, Santa Olalla and Las Pajas lagoons versus monthly precipitation in the study area.

The more likely hypothesis (yet to be confirmed) to explain this increase in SO_4 is the oxidation of sulphur compounds from the lagoon-bottom sediments and/or the dissolution, at the beginning of the wet season, of sulphate salts precipitated in the swampy shore lagoons during the dry period. Afterwards these surface waters increase their $\delta^{18}O$ content by evaporation while SO_4 gradually disappears by reduction and incorporation into the lagoon-bottom sediments. As yet, sulphur and sulphate isotope studies have not been carried out.

CONCLUSIONS

From the chemical point of view, three groundwater groups can be differentiated: i) a Na-Cl type groundwater, located mainly in the silicious Eolian Unit, which is representative of local recharge water; it is slightly concentrated with respect to the atmospheric contribution; this groundwater flows through almost totally leached siliceous sediments, so that the initial chemical composition of groundwater is barely altered by the flow through these materials; ii) groundwater of Na-Ca-Cl-HCO_3 type that belong to the Deltaic Unit; some sediment layers containing carbonated debris provide Ca and HCO_3 to the initial composition of the water; and iii) a group formed by different types of groundwater interacting with surface waters from the lagoon complexes, the last being affected by evaporation, sulphur oxidation and gypsum-alternated precipitation and dissolution in the swampy lagoon area.

According to the estable isotopes, the groundwater of the area shows the isotopic signature of local rainfall. The isotopic composition of the lagoon waters is also controlled by the rain and is greatly affected by surface evaporation and the inflow and outflow of groundwater. At the same time, it depends on the lagoon flood level. When the volume of lagoon water decreases in the dry season it becomes more concentrated and heavier in $\delta^{18}O$ and $\delta^{2}H$; when the volume increases, a dilution takes place making the lagoon waters lighter and simultaneously the SO_4 content increases.

The temporal evolution of lagoon water chemistry is controlled by the rain, the evaporation cycles, the local inflow and outflow of groundwater and the chemical processes taking place both in the water column and in the lagoon-bottom sediments. The chemical composition of some of the studied lagoons is linked to the degree of flooding; the SO_4 concentration in the water increases when the lagoon level is higher. This process could be due to SO_4 reduction in the summer, sulphur accumulation in the bottom sediments and subsequent oxidation by rainfall in the wet season and/or the dissolution of precipitated sulphate salts in the swampy lagoon margins in the drying up period of the dry season.

ACKNOWLEDGEMENTS

This work is the continuation of several projects within the Doñana area: Doñana (PB 87-0842, AMB 92-636, AMB 95-0372) and has been carried out in the framework of several others already finished or still ongoing: MADRE I and MADRE II (CICYT HID-97-0321-C02 and CICYT REN 2001-1293-CO2-O2/HID carried out in cooperation by the Autonomous University of Madrid and the Technical University of Catalonia at Barcelona, Spain); BASELINE (EUK1-CT1999-0006) and 'Bromides' (HID 1999-0205). The authors would like to thank the CHG (Guadalquivir River Basin Authority) for their extraordinary support with the fieldwork and the IGME (Geological Survey of Spain) for the analysis of major elements and the field support of geologist, Carlos Mediavilla.

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CHAPTER 19

Alluvial aquifers at geological boundaries: Geophysical investigations and groundwater resources

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ABSTRACT: Alluvial channel sands in active ephemeral streams are potentially highly productive aquifers that are normally fully recharged annually. The groundwater resource is constrained by the limited three-dimensional extent of these aquifers. Concepts are developed that propose an increase in alluvial aquifer dimensions at geological boundaries. Multi-electrode resistivity and ground penetrating radar are used to investigate the dimensions of an alluvial channel at a geological boundary with the more resistant lithology upstream. These investigations reveal that alluvial channel fill dimensions are increased in the overlying less-resistant lithology downstream of the boundary. Groundwater flow modelling has been used to determine aquifer potential and identify key fluxes, indicating that significant irrigation potential exists from these aquifers.

INTRODUCTION

Alluvial aquifers are widely used in the semi-arid regions of southern Africa, both for primary water supply and for irrigation development (Thomas & Hyde, 1972; Wikner, 1980; Nord, 1985; Owen & Rydzewski, 1991). These aquifers occur in the channels of active ephemeral rivers, sometimes called ‘sand rivers’ since they appear as dry sandy river beds during the dry season after river flow has ceased. Significant commercial irrigation development has taken place from the alluvial sediments of the Umzingwane, Runde and Limpopo rivers in southern Zimbabwe. This irrigated production coexists side by side and in contrast with low productivity dry land farming as practised by the local communal farmers. At Mazunga ranch on the River Umzingwane, an 800 ha block of commercial irrigation is developed by means of a suction lift well point system driven into alluvial channel sands upstream from a geological contact between resistant silicified sandstone downstream and less competent basalt upstream (Ekström et al., 1996). The neighbouring communal farmers live alongside the same river in abject poverty, practising dry land cultivation to try and provide for their basic food requirements.

This research has been carried out to determine whether there exists sufficient groundwater within this and other similar alluvial systems to provide for irrigation development capable of supporting the basic needs and improving the livelihoods of the communal farmers in these areas.

The key factors that govern the magnitude of exploitable groundwater resources in such alluvial aquifers in southern Africa have been identified and are listed below.

- In semi-arid climates, alluvial aquifers are totally recharged annually due to indirect recharge through river bed infiltration and may therefore be fully exploited on an annual basis. It has been shown that river flow only occurs after the aquifer channel sands have become fully saturated (Nord, 1985; Halcrow, 1982). Rivers are essentially natural rainwater harvesting systems, collecting run-off from large catchment areas and channelling this into narrow bounded river channels. The investigated site on the River Umzingwane has an upstream catchment of approximately 18,000 km^2.
- Aquifer dimensions are determined by the extent and thickness of the alluvial fill in the river channel and under the lateral alluvial plains, where these are developed. In general, alluvial aquifer dimensions are of the order of a few tens to a few hundreds of meters in width and a few to a few tens of meters in thickness, developed along the length of the alluvial channel, resulting in a long thin ribbon like aquifer. The geometrical extent of the saturated alluvial fill is a key limiting factor.
- The alluvial aquifer channel sediments are generally clean washed sands that have excellent hydraulic and storage characteristics, allowing for high well pumping rates. Hydraulic conductivity values ranging from 100–250 m/day and specific yield values ranging from 10–35 per cent have been recorded in Botswana and Zimbabwe (Nord, 1985; Owen, 1994).
- The flow through the channel sediments from upstream and downstream into the well field abstraction zone is identified as a critical factor controlling the productivity of the well field.
- Complete annual recharge ensures generally good fresh water quality.

Based on the foregoing, it can be seen that the three-dimensional extent of the saturated aquifer represents a key constraint for the development of groundwater resources from alluvial systems of this type. Localities where the alluvial fill dimensions have been naturally enhanced represent suitable sites for the optimum development of the groundwater resource in these aquifers. The geological map of Zimbabwe (1:1,000,000) shows that alluvial deposits tend to occur preferentially at geological boundaries (Owen, 1994). Knowledge of the bedrock geology and an understanding of the processes of alluvial deposition are helpful for determining localities with potentially enhanced alluvial aquifer dimensions. Such localities may host groundwater resources capable of supporting large-scale water development such as is required for commercial irrigation.

This paper focuses on the geometry of the alluvial fill at a geological boundary and models the groundwater resource potential of the associated alluvial aquifer. The data presented comes from a case study on the River Umzingwane in southern Zimbabwe (Figure 1).

DEVELOPMENT OF ALLUVIAL AQUIFER FILL AT A GEOLOGICAL BOUNDARY

Alluvial aquifer fill is deposited during periods of aggradation, due to a change in river flow regime or an increase in sediment supply to the stream (Richards, 1982). The alluvial sediments cover and fill the pre-existing river bed topography, normally giving rise to a planar gently sloping sand bed.

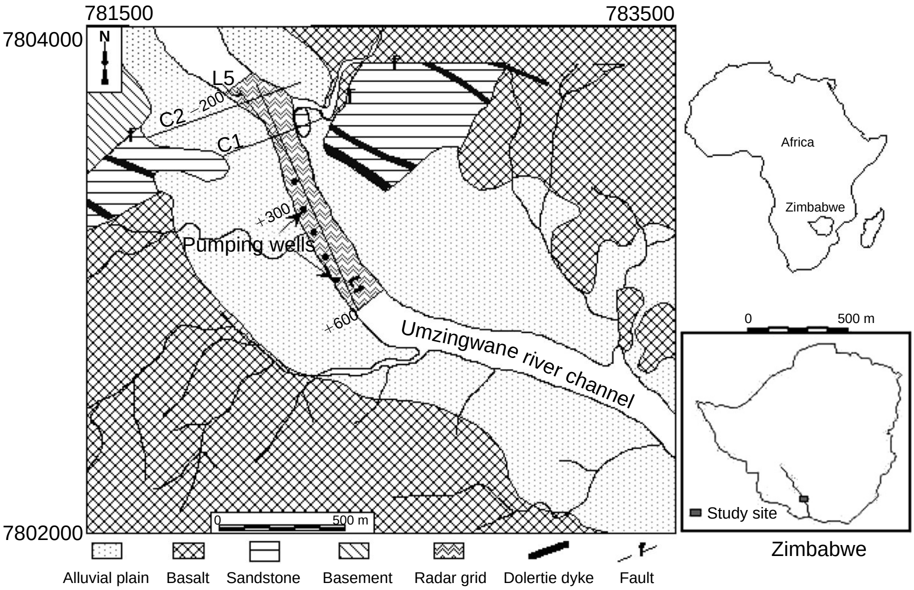

Figure 1. Geological map of the Umzingwane River site at Bwaemura. The positions of the resistivity profiles C1, C2, L1 and L5, and the area covered by the radar grid are shown.

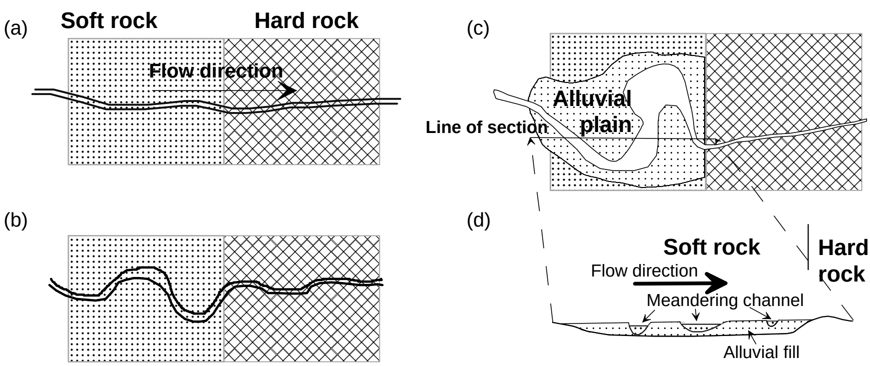

Figure 2. Alluvial development: resistant lithology downstream. Plan view: a) River channel cuts a narrow gorge through the hard rock downstream of boundary. b) Meanders develop upstream of contact due to barrier effect of resistant downstream rock combined with erodibility of upstream soft rock. c) Meandering upstream of contact produces wide shallow alluvial valley, which is filled with alluvial sediments during periods of aggradation. Cross-sectional view: d) Channel meanders cut the shallow alluvial fill in the alluvial plain.

It is proposed that alluvial fill is augmented, in width and in thickness, at geological contacts. The effect of geological boundaries on the geometry and position of the alluvial fill depends on whether the resistant lithology occurs upstream or downstream. In the case with a downstream resistant rock barrier, a shallow meandering river channel occurs upstream of the geological boundary. A subsequent period of alluvial accumulation will give rise to a shallow laterally extensive alluvial fill, consisting of both channel and alluvial plain fill materials, as shown in Figure 2.

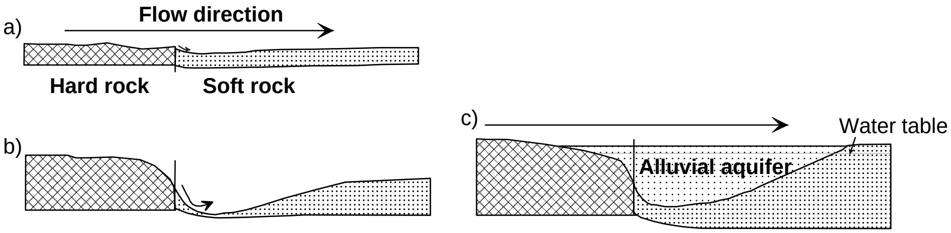

Figure 3. Alluvial development: less-resistant lithology downstream. Cross-sectional view: a) Flow over the boundary results in differential erosion; a small depression forms in the softer rock. Subsequent flow causes eddy currents, which scour out the softer rock. b) Further scour results in the development of a waterfall. c) Flow recession after flooding can result in alluvial aggradation, which fills up the channel, giving rise to an increased alluvial fill thickness downstream of the boundary.

In the case with the more resistant geology located upstream, a waterfall is commonly developed by scour of the more easily eroded downstream lithology. If there is a subsequent period of alluvial aggradation, the waterfall and associated downstream plunge pool are buried, thus becoming the locus of enhanced alluvial fill thickness as shown in Figure 3.

Such conceptual models, which illustrate the deposition and accumulation of alluvial fill material, may be used as a guide for locating sites in sand rivers where greater volumes of alluvial fill can be expected to occur. In this study, only one alluvial aquifer site is investigated. This site has the more resistant geology upstream and a significant increase in the saturated thickness of the alluvial fill downstream of the geological boundary was anticipated.

These 'buried waterfall'-type sites are considered to be more favourable for groundwater development than the laterally extensive sites, due to greater saturated thickness of the alluvial fill, increased available drawdown for wells and reduced water losses to evaporation.

RIVER UMIZINGWANE ALLUVIAL AQUIFER

The alluvial fill at a geological boundary (Figure 1) along the River Umzingwane in southern Zimbabwe was investigated using multi-electrode resistivity and ground penetrating radar, in order to obtain information on the bedrock profile and on the dimensions of the alluvial fill. There are two boundaries in close proximity along the river channel at the selected locality. Gneiss occurs in the northern upstream reach of the channel in faulted contact with the downstream Karoo sandstone, which is heavily silicified within the fault zone. The sandstone dips gently to the south and is overlaid by Karoo basalt downstream. Although the basalt normally overlies the sandstone conformably, the contact between the two at this locality has been mapped as an inferred fault.

Based on outcrop evidence at this locality, it is suggested that the gneiss and silicified sandstone in the fault zone are more resistant to erosion and weathering than the sandstone downstream, which in turn appears marginally more resistant than the downstream basalt. With regard to the concepts of alluvial fill development proposed in the previous section, both boundaries have weaker rock downstream, and deeper alluvial sections may

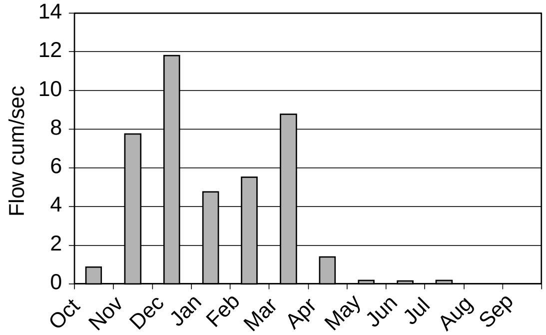

Figure 4. Mean monthly river flow records for Kwalu gauging station, Umzingwane River – average of 7 years of available flow records.

be expected to occur at and just downstream of these boundaries. However, the major contrast in competence is between the upstream gneiss/silicified sandstone fault zone and the downstream sandstone, and a more significant enhancement of the alluvial fill was expected just downstream of this boundary.

For the alluvial channel sands, aquifer parameter values for hydraulic conductivity, specific yield and porosity were measured using a constant head permeameter and by laboratory gravimetric measurements respectively. Based on these measured values and values obtained from other alluvial channels in the region (Nord, 1985; Owen, 1994), average hydraulic parameter values have been assigned to the alluvial channel sediments as follows: hydraulic conductivity: 200 m/day; specific yield: 20 per cent; and porosity: 35 per cent.

As indicated previously, recharge to semi-arid alluvial aquifers is completely dominated by river flow. A gauging weir exists 7 km downstream from the investigated aquifer and the flow data indicate that there is river flow greater than 1 m^3/sec for 6–7 months each year (Figure 4). It can therefore be assumed that the aquifer is fully saturated for this period, provided that abstraction rates from the aquifer are less than the measured river flow. Depletion of the alluvial groundwater resource only begins to occur once river flow has ceased (Nord, 1985; Halcrow, 1982).

GEOPHYSICAL INVESTIGATIONS

Initially, electrical resistivity profiling was used to determine the subsurface structure at the study site. The field data were collected by Ekström et al. (1996) as continuous vertical electrical soundings (CVES) using the ABEM Lund Imaging System (Dahlin, 1996) with the Wenner array selected in order to depict the anticipated horizontal nature of the alluvium/bedrock boundary. Protocol files with ten different electrode spacings in the range 5–120 m were used to provide a compromise between time required for field data acquisition, depth penetration and resolution of the shallow subsurface environment. The resulting apparent resistivity sections were processed by means of inverse numerical modelling to provide estimates of the true resistivity distribution, using the software Res2dinv (Loke, 1999).

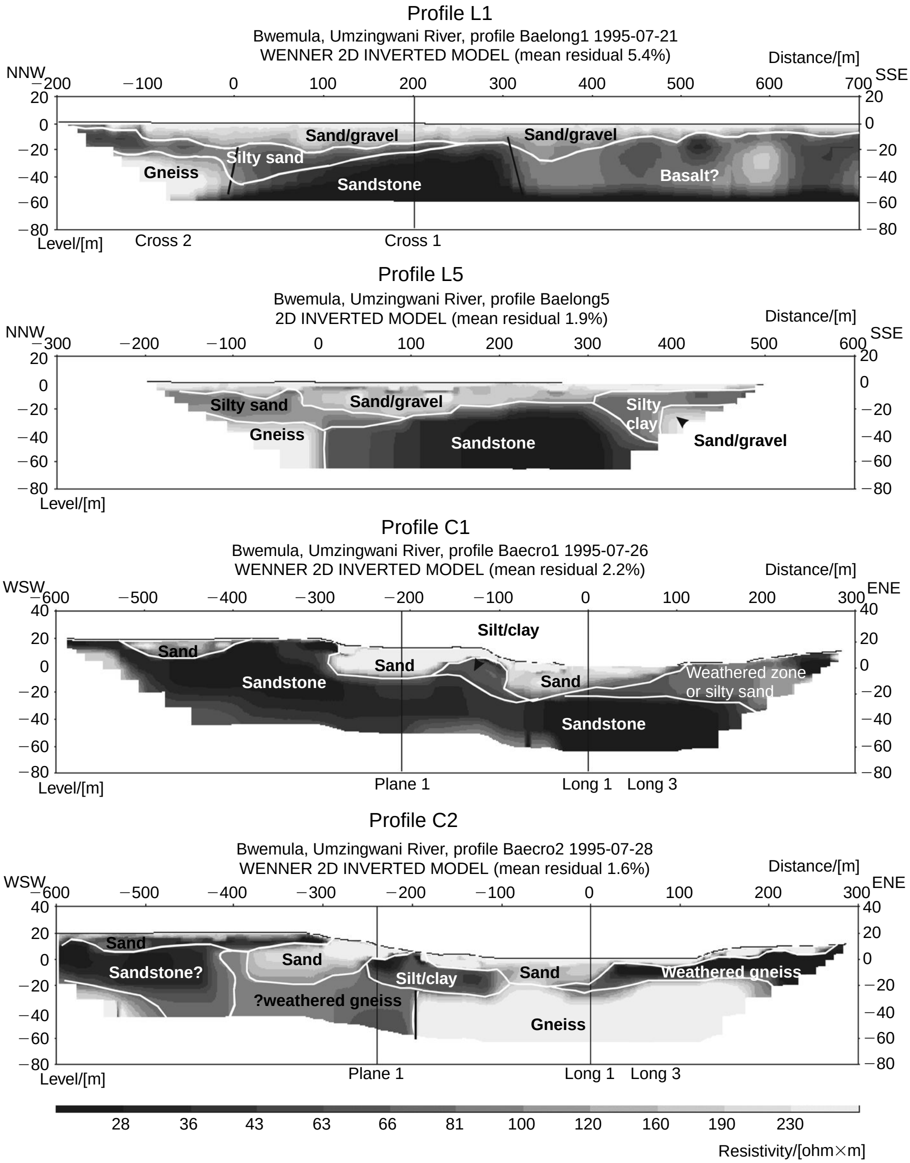

Figure 5. Multi-electrode resistivity profiles across the contact zones. L1 and L5 are profiles along the river and C1 and C2 are profiles across the river. Alluvial channel fill has a high resistivity signature and this helps to distinguish it from the other earth materials in the section. The interpreted geology is marked on to the sections.

Four of the measured CVES profiles have been selected to illustrate the effect of the observed geological boundaries on the geometry of the alluvial fill (Figure 5). Profiles L1 and L5 were measured along the length of the channel from north-west to south-east and positioned so as to intersect the geological boundaries, while profiles C1 and C2 were measured across the channel and extended to cover the full width of the lateral alluvial plains (Figure 1). Figure 5 shows the inverted electrical resistivity sections of all these profiles with the interpreted geology marked directly on to the images.

The alluvial channel fill has a high resistivity signature and this makes it relatively easy to distinguish from underlying bedrock, except where the bedrock is fresh gneiss, which is also highly resistive. Profile L1 shows an increase in alluvial fill thickness between 0–150 m, in the vicinity of the gneiss/sandstone boundary, and again at 300–400 m, at the sandstone/basalt boundary. Profile L5 shows a similar pattern, but the thickening of the alluvial fill downstream of the gneiss/sandstone boundary at 0 m is less striking. Profiles C1 and C2 cross the channel and clearly show the presence of a paleo-channel at a higher elevation and to the west of the active channel.

The maximum thickness of alluvium occurs at the gneiss/sandstone contact (0 m on L1), where the alluvium appears to attain a thickness up to 40 m. The thickness of the alluvial fill declines in a downstream direction moving away from the geological boundary until it remains only as a thin (~5 m) surficial layer.

In addition to the resistivity profiles, Beckman and Liberg (1997) carried out a ground penetrating radar survey across the geological boundary area. A radar grid was surveyed within the active river channel where good penetration was expected. The grid extends across the full channel width and along the same length of channel as the resistivity line L1 (Figure 1). It consists of eight parallel lines 25 m apart, aligned along the channel and nine parallel lines across the channel approximately 100 m apart. The radar grid was not extended on to the alluvial plains due to poor access as a result of the dense riverine vegetation in these areas. In addition, there was concern that radar signals would suffer from severe attenuation in silts and clays that can be expected as over-bank deposits on alluvial plains.

The data were collected with a Malå Geoscience RAMAC/GPR, using antennas with a centre frequency of 50 MHz. The distance between each trace was 20 cm. The radar results were plotted as standard radargrams, where the depth axis shows reflection time. True depths were estimated from WARR (wide angle refraction and reflection) measurements at four selected points. The primary reflector was digitized from the radargrams and the digitized depths merged into a three-dimensional depth map (Figure 6), which has the same grid coordinates as the resistivity profiles (Figure 5).

The three-dimensional radar image (Figure 6) shows a reflector underlying the alluvial channel sediments, which is interpreted as bedrock beneath the alluvial fill. The maximum depth of penetration attained by the radar survey was 16 m and the reflector is lost as the bedrock depth increases towards the western side of the channel. There is no bedrock reflector on the western edge of the alluvial channel (Figure 6), suggesting that a paleo-channel extends beyond the western edge of the active channel, an interpretation that is corroborated by resistivity profiles C1 and C2 (Figure 5).

The lateral extent of this buried paleo-channel can be estimated from the vegetation signature on the alluvial plains (Figure 6). Aerial images show a strongly bimodal vegetation signature along the banks of the river, consisting of a belt of lush thick vegetation with large riverine trees such as *Acacia galpinii* and fig species such as *Ficus sycomorus* occurring closer to the channel, and a more sparse open woodland with smaller non-riverine trees occurring further away from the channel. The lush vegetation signature on the alluvial plain is indicative of the availability of a perennial water supply to the root zone and is interpreted as saturated alluvial fill in the buried paleo-channel, which is in hydraulic contact with and is recharged from the active river channel.

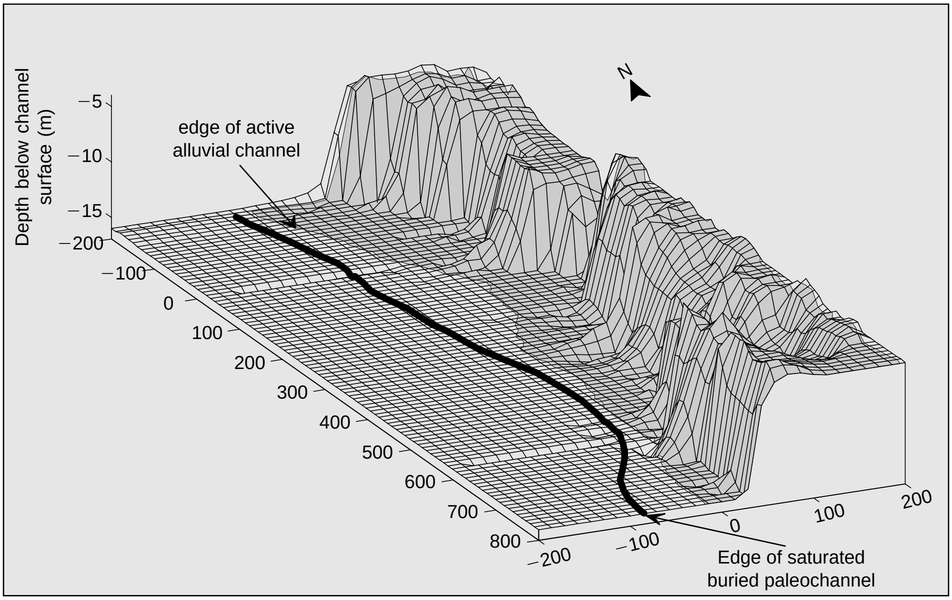

Figure 6. Three-dimensional radar image of the Umzingwane River channel. The image shows the bedrock surface beneath the alluvial channel. The riverbed surface is at 0 m on the vertical scale in the figure. The maximum penetration depth attained was 16 m, which is insufficient to locate bedrock in the western portion of the channel. The edge of the saturated alluvial fill to the west (solid dark line) has been identified from the riverine vegetation signature on satellite images. The position of the grey radar grid is shown in Figure 1.

EFFECT OF GEOLOGICAL BOUNDARIES ON ALLUVIAL FILL DIMENSIONS

The site geological map (Figure 1) indicates that there has been dextral displacement of the gneiss/sandstone boundary beneath the alluvial fill in the Umzingwane valley. As a result of this displacement, it is difficult to determine from the map the exact position of the geological boundaries where they are obscured by the alluvial cover. The radar grid (Figure 6) has been divided into two equal sections: the northern section from −200 m to +300 m and the southern section from +300 m to +800 m. The map (Figure 1) shows that the northern radar section extends across the gneiss/sandstone boundary. However, due to this dextral displacement, the position of the sandstone/basalt boundary beneath the alluvial channel cover is not clear from the map, but the resistivity sections L1 and L5 indicate that this boundary occurs at approximately +300 m. The southern section of the radar grid from +300 to +800 m therefore does not cross any geological boundary, but has a low competence contrast geological boundary at the start of the section.

The three-dimensional radar image (Figure 6) shows that the alluvial fill dimensions are both wider and deeper in the northern portion of the channel, as compared to the southern channel section, which is largely filled with bedrock. The relative dimensions of the alluvial fill have been estimated from the radar image. If the bedrock reflector persists westwards at 16 m depth to the edge of the buried paleo-channel (Figure 6), then the upstream northern section comprises 65 per cent and the downstream southern section 35 per cent of the aquifer. If the bedrock reflector were to rise to the surface just at the edge of the active channel, then the upstream sector would comprise 64 per cent and the downstream sector 36 per cent of the aquifer. These values support the proposal (Figures 2 and 3) that alluvial fill volumes are significantly increased at geological boundaries.

GROUNDWATER MODEL

Groundwater flow modelling has been used as a guide for determining the various fluxes into and out of the model domain, and the optimum pumping rate and well field spacing for an alluvial aquifer of the type described here.

The aquifer is modelled as a single layer, 200 m wide and 1000 m long and consists of 20 columns and 50 rows with uniform grid block dimensions of 10×20 m. The aquifer dimensions for the active channel are taken directly from the three-dimensional radar image (Figure 6). The saturated alluvial fill in the buried paleo-channel west of the active channel is not included in the model, since its lateral extent is only inferred from the vegetation signature on aerial images and its thickness is not known. The model therefore underestimates the groundwater resource in the aquifer, potentially by as much as 40 per cent, and the values derived from the modelling may be treated as the minimum aquifer potential for this section of river. The model design presumes no fluxes occur through the lower (alluvium/bedrock) boundary.

The assigned aquifer parameter values for hydraulic conductivity and storage are discussed in a preceding section. These values remain unchanged for all the modelled scenarios.

There are potentially major fluxes into and out of the model space through the river bed boundary and through the upstream and downstream boundaries. Recharge to the aquifer occurs mainly from river bed infiltration, which predominates over the direct rainfall recharge. This indirect recharge is modelled as a river boundary, with the water level elevation above the channel surface during periods of river flow. Once the river flow ceases during the dry season, the river boundary is turned off. Direct recharge is estimated at 10 per cent of average annual precipitation (55 mm) for all scenarios and occurs entirely during the period river flow.

This river boundary is a transient boundary covering the entire model surface, set with the river stage elevation 0.5 m above the channel surface for 180 days, and then turned off for the remainder of the year. To allow for water transfer from the river into the aquifer, a conductance value of 500 m^2/day has been assigned to all cells in the river bed. The boundary and conductance values for all the modelled scenarios do not at all limit the required inflows during the period of river flow. Drawdown within the aquifer only occurs after the end of period of river flow and this is the critical stress period for the groundwater resource.

In addition to the flux through the river bed, water may also enter or leave the model through the upstream and downstream boundaries, which are designated as general head boundary cells. The flow through these boundary cells is controlled by the conductivity and saturated aquifer thickness in the boundary cells. Figure 6 shows that the western side of the aquifer has greater depth (15 m) of alluvial fill than the eastern side (5 m) and this is reflected in the assigned conductance values in the boundary cells on the west and east sides of the model, which are set at 1500 and 500 m^2/day respectively.

Table 1. Assigned head/time and conductance values for general head boundaries.

Time start	Time end	Scenario 1 (Low head m)	Scenario 2 (Med head m)	Scenario 3 (High head m)	16 m thick (Cond m^2/d)	5 m thick (Cond m^2/d)
0	180	0	0	0	1500	500
180	210	−1	−0.5	−1	1500	500
210	240	−2	−1	−1	1500	500
240	280	−3	−1.5	−1	1500	500
280	320	−4	−2	−1	1500	500
320	365	−5	−2.5	−1	1500	500

The saturated thickness at the boundary cells will decline as a result of pumping, evaporation and other outflows from the aquifer. The actual saturated thickness is not known and to cover the possible range of boundary cell head values, four different scenarios have been modelled; zero flux with the general head boundary turned off, low flux with a small saturated thickness assigned to the boundary cells, moderate flux with a medium saturated thickness assigned to the boundary cells and high flux with the boundary cells fully saturated except for a dry top 1 m, which is annually lost to evaporation (Wipplinger, 1958). Transient boundary heads are assigned in all model scenarios and are set at river bed level during the period of river flow and decline with time during the period of no flow. For all assigned boundary head values, the model has been adjusted for maximum pumping discharge.

Table 1 shows the general head boundary input values for time, head and conductance for all the modelled scenarios. The first time period, 0−180 days, represents the period of river flow (Figure 4) and during this time period the head values at the boundary cells are set at 0 m, which is the river bed surface. Subsequently, the head values are lowered by equal increments of either 0.5 m or 1.0 m for each time period for the medium and low head scenarios respectively as shown in Table 1. The time periods are arbitrarily set at 40 days each, except for the two time periods immediately after the end of river flow (day 180−210 and day 210−240), which are set at 30 days each. The rationale for assigning shorter time interval per fixed drawdown in the 60 day period immediately after the end of river flow is that during this period head changes are subject to an additional outflow flux due to evaporation losses as well as the pumping and general head boundary fluxes, which persist throughout the hydrological year. For the high head case, only two time intervals have been used: the period of river flow with the head set at 0 m and the period of no river flow with the head set at −1.0 m.

Six pumping wells have been placed in the model (Figures 1 and 7) and the pumping rate from these wells has been increased incrementally to obtain the maximum discharge for each model scenario.

MODEL RESULTS

The model flow budget (Table 2) shows that during the period of river flow (day 1−180), inflow through the river bed boundary completely dominates the inflow water budget and provides water not only for the pumping wells, but also surplus water which flows out of the aquifer via the upstream and downstream general head boundaries. As the general

Table 2. Water Budget for alluvial aquifer models.

Scenario ↓	Period of river flow (0–180 days)					Period of no river flow (180–365 days)				
	Inflow		Outflow			Inflow (average)		Outflow (average)		
Flux boundary	River	Rechg	Wells	G.h.b.	E.t.	Storage	G.h.b.	Wells	G.h.b.	E.t.
Zero flux	3267	55	2500	0	822	2500	0	2500	0	0
Low flux	10471	55	3500	6204	822	1154	2482	3500	71	64
Medium flux	13377	55	7500	5110	822	909	6628	7500	0	37
High flux	15870	55	10000	5103	822	28	9997	10000	0	25

Rechg = recharge, G.h.b = general head boundary and E.t. = evapotranspiration. The average values for the period of no river flow reflect the transient modelled head changes at the general head boundaries.

head boundaries are adjusted to allow for greater fluxes, pumping rates are increased and these increases are met by increased inflow through the riverbed boundary. For all scenarios with active general head boundaries, the outflow flux remains reasonably steady, which presumably is conditional on the number and position of pumping wells. The fact that these outflow fluxes are so similar suggests that the maximum pumping rate has been attained for these model scenarios. For the period of river flow, model convergence occurs readily.

The model period of no river flow (day 180−365) is less robust and in many cases well discharge had to be reduced in order to achieve model convergence.

The model indicates that the key outflow is to pumping wells and that this demand is balanced by a combination of inflow across the general head boundaries and water taken from storage within the modelled aquifer. As the boundary conditions are adjusted to allow for higher fluxes, so the quantity of water taken from storage declines, while the inflow across the boundaries increases. In the case with the high flux boundary, very little water is taken from storage and all the water required for pumping enters the aquifer through the general head boundaries. The values of the various fluxes for the different boundary conditions are summarized in Table 2. For the medium flux scenario at time 365 days, plan and cross-section views (Figure 7a) of the channel show water levels in the aquifer, dry cells and velocity vectors converging towards the wells. Drawdown levels within the aquifer range from 2−8 m and groundwater flows occur through the general head boundaries at the up and downstream ends of the modelled channel.

The zero flux boundary may be regarded as a special artificial case, where no water is allowed to enter the aquifer during the period of no river flow (day 180−365). The pumping rate has been increased incrementally until no further model convergence can be obtained. Since there is no inflow or outflow from the zero flux model, the model results provide information about the quantity of water that can be pumped from available storage in the model domain. The total volume pumped in the dry season is 462,500 m^3, which is considered to be the maximum exploitable groundwater stored in a 1-km river reach, without any inflow, as may be the case in periods of extreme drought. Provided that the channel is fully saturated at the start, a maximum pumping rate of 2500 m^3/day can be sustained for the entire dry season. For the zero flux scenario, Figure 7b illustrates plan and cross-section views at 365 days of the channel showing dry cells where the aquifer is thinner and drawdown of 9−11 m within the deeper aquifer zones. Comparing Figures 7a and b provides a view of the importance of the fluxes into

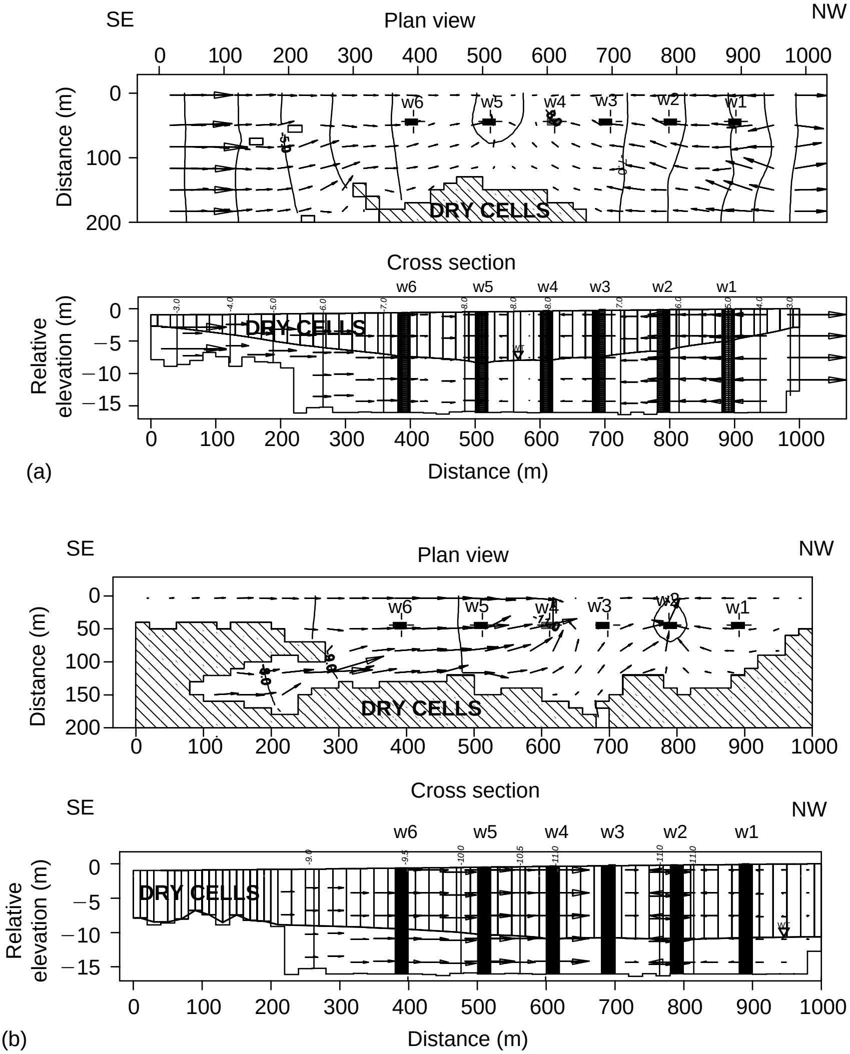

Figure 7. Plan and cross-section (column 5) views of aquifer showing location of pumping wells, water levels, dry cells and velocity vectors at 365 days for: a) intermediate drawdown medium flux general head boundary and b) zero flux boundary condition.

the aquifer through the general head boundaries. These inflows reduce the drawdown in the aquifer and allow for higher pumping rates. Such information may be used for optimization of the well field design.

IRRIGATION POTENTIAL OF ALLUVIAL AQUIFERS

If the average daily irrigation requirement is arbitrarily set at 50 m^3/day/ha, then a pumping rate of 2500 m^3/day/km reach of river should be sufficient to irrigate 50 ha. If pumping rates are increased, this will deplete the water resource in the aquifer upstream and downstream. Using six pumping wells a modelled pumping rate of 10,000 m^3/day, sufficient to irrigate 200 ha, is sustainable for the 185-day period of no river flow and would deplete the water in storage from a 4-km river reach of the aquifer. Based on this, it is clear that the Umzingwane alluvial aquifer is highly under-utilized and that irrigation could be developed far more extensively from the alluvial sediments within the river channel.

CONCLUSIONS

A conceptual geological model that supports a significant increase in the geometrical extent of alluvial aquifers at geological boundaries is suggested. Geological maps and geological site investigations support this model. Geophysical surveying confirms increased thickness and lateral extent of alluvial channel sediments at faulted geological boundaries in the Umzingwane River in southern Zimbabwe. Such channel sediments constitute an excellent aquifer, which is generally fully recharged every year.

The employed geophysical techniques proved suitable for mapping the alluvial aquifer. Resistivity imaging gave valuable results inside the present-day stream channel as well as the flood plains and surrounding terrain. GPR only yielded useful results within the coarse channel sediments, with a maximum depth penetration of 16 m and depth penetration was reduced by clay or silt lenses. The geophysical results were used to delineate the three-dimensional geometric extent of the aquifer used for designing the groundwater model space. The geophysical data would provide excellent guidance for drilling.

The groundwater model indicates that the Umzingwane alluvial aquifer is under-utilized and that significant irrigation potential exists. These results are expected to be applicable to similar river channels, which could presumably provide irrigation water to alleviate both the risks and the low yields associated with dry land farming in similar semi-arid areas, both within Zimbabwe and elsewhere.

ACKNOWLEDGEMENTS

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CHAPTER 20

Evolution of groundwater protection policies in developing cities: Stakeholder consultation case studies in Bangladesh and Kyrghyzstan

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ABSTRACT: The pressure of rapidly expanding urban populations in groundwater-dependent cities in the developing world is a major groundwater management issue. City planners in emergent economies realize that water resources are finite but wrestle with the need to introduce sustainability into their plans for infrastructural improvement without unnecessarily constricting economic expansion by overly restrictive policies. Realistic considerations of enforcement ability make it vital to involve urban groundwater stakeholders as early as possible in policy development if urban aquifer protection is to stand any chance of success. The experience of two such developing cities in Bangladesh and Kyrghyzstan is described where, despite limited resources and knowledge, base attempts are being made to develop a locally appropriate and acceptable groundwater protection plan along sound hydrogeological principles. In both cities, the novel exercise of stakeholder consultation was considered by participants to be a welcome innovation.

NEED FOR URBAN AQUIFERS TO BE MANAGED FOR CONSERVATION

Global trends in urbanization mean that by 2010 it is estimated that half the world's predicted population of 6500 million will live in towns or cities (UNCHS, 1987). Between 1980 and 2000, 85 per cent of urban population growth was concentrated in the developing world, such that by the year 2000, about twice as many people were living in cities in developing countries (1900 million) as in the developed world (950 million). Fair access to water supply and sanitation has always been a key issue in expanding cities, but the sheer scale and extent of global urbanization is placing unprecedented pressure on regional water resources around urban agglomerations (UNEP, 1996).

The ability to replace a degraded urban aquifer by water tapped from the hinterland is a fast-receding luxury because increasingly such catchment resources may already be fully utilized for agricultural or ecological purposes (Burke & Moench, 2000). Coupled with moves in the spirit of the 1992 Earth Summit manifesto (signed by more than 150 countries), demands are increasing to introduce sustainability principles into new water supply projects. In future, a city fortunate enough to possess a significant urban aquifer resource will no longer be able to assume that it is a discardable asset that can just be abandoned once depleted or heavily contaminated. Added to the issue of resource equity is the growing realization that the water infrastructure of a city and an underlying aquifer system are interdependent. Mismanagement, or more commonly absence of management, can result in the same city experiencing drastically falling water levels during early expansion followed by groundwater flooding at later development stages (Morris et al., 1997) or incurring unforeseen and expensive future treatment costs to counteract the results of persistent contamination due to poorly planned/controlled activities at the land surface (Ahmed et al., 1998; Seddique, 1998).

It is thus a major urban challenge to introduce not only aquifer protection principles into municipal planning but also to engender a greater ownership of such policies by all users of the urban subsurface in order to make them enforceable. This paper describes results from an ongoing collaborative research project[1] that seeks to demonstrate how such principles and a corresponding sense of ownership can be introduced into the dynamic but frequently impoverished world of small but fast expanding cities.

BACKGROUND TO THE PROJECT

The project commenced in late 1998 in the two developing cities of Narayanganj, Bangladesh and Bishkek, Kyrghyzstan with the following objectives:

- to employ available data to conduct aquifer vulnerability and subsurface contaminant load surveys to provide pollution risk assessments in each case study city;
- to use these assessments to engage groundwater stakeholders in the development of policy options for a city groundwater protection plan comprising a concise set of policy guidelines and a groundwater resource-planning map;
- to generalize the lessons learnt from the case studies for wider use by other groundwater-dependent developing cities.

The rationale of the project is to demonstrate whether practical aquifer protection policies can be developed using the urban groundwater management prescriptions espoused by the World Bank (Foster et al., 1997), yet remaining within the limited financial/institutional resources typically available to managers and planners of the water infrastructure of an emerging-nation groundwater-dependent city.

The first objective of the project (pollution risk assessment) has been reported on in Morris et al. (2001) and this paper describes the stakeholder consultation process.

URBAN PROTECTION POLICY CONSIDERATIONS

Why urban aquifer protection plans are uncommon

Globally, despite the widespread acceptance, in the abstract, of the benefits of groundwater protection, urban aquifer protection plans are, in practice, still unusual. This has been ascribed (Morris et al., 1997) to:

- **Inability to see 'the big picture':** Wells may be drilled by a single utility or by any of hundreds of private users, fragmenting the knowledge base; problems such as overdraft or water quality deterioration are thus less easy to identify in their early stages; borehole construction is usually incremental and needs relatively low investment levels so it is less likely to be the subject of a city master water plan.

- **Sustainability linkage unrecognized:** For the general public, reaction/residence times of water in observable features such as rivers are much easier to grasp than groundwater timescales; the strong sustainability focus of a resource whose pollution response and replenishment timescales are typically measured in years → decades → centuries rather than hours → days → weeks is not so widely appreciated. The political outcome of this is 'out of sight, out of mind'.
- **Lack of data obscures 'the clear picture':** Many urban water databases are not consciously linked to a management need, particularly early-warning surveillance. So when aquifer assessment is undertaken the results are often highly qualified because much laboriously collected information is found to be either inapplicable or inappropriate; such qualifications are necessary but hinder policymaking decisions.

Therefore, despite the high dependence of both case-study cities on local groundwater, it was not at all surprising that neither had an aquifer protection strategy and that in effect, groundwater had been developed opportunistically in each case.

Need for pragmatic design criteria

Any set of aquifer protection policies to be applied to an already-existing urban area will need to evolve strategies which, while they constrain land use, accept trade-offs between competing interests and utilize the natural contaminant attenuation capacity of the strata overlying aquifers (Foster & Skinner, 1995). To implement such strategies hydrogeological understanding needs to inform land-use policies and provide simple robust matrices that indicate what activities are possible where, at an acceptable risk to groundwater. In turn, construction of such matrices requires pragmatic design criteria if planning is not to be so delayed as to irretrievably prejudice resource sustainability. Such criteria, which need to be targeted *from the outset* for a subsequent policy development and aquifer management stage, include:

- **Uses available data:** The typical situation would be that projects of this type would be resourced only to use existing data, i.e. either basic data arrays already collected for other purposes or simple parameters easily collated from operational records. In Narayanganj, the standard of basic hydrogeological data was relatively poor, being limited to a handful of borehole logs in the centre of the city. In Bishkek, the standard of basic hydrogeological data was good, being comprehensive in the parameters covered (geology, hydrogeology, water levels, location of wells, etc.), internally consistent and relatively up to date (mostly less than 20 years old). In both cities however routine monitoring information was poor or out of date, so that trends in aquifer usage and water quality were unknown.
- **Employs transparent tools:** To facilitate wide uptake, the tools used need to be simple and robust so they can be generalized to many different city situations with relatively little modification. The process needs to be relatively rapid, low cost and easy to undertake with limited human, technical and financial resources. For example, while digital GIS techniques were used in these case studies to permit easy overlay of thematic material for map production, the number of stages was small enough and the

ranking system simple enough for manual overlay techniques to have been employed if local resources had so dictated. A corollary is that the use of now widely available GIS software packages should not obscure the quality (or sparseness) of underlying data. Even more importantly, where it proves impossible to avoid using qualitative (but still objective and rational) techniques, these need to be openly but not deprecatingly described as such, in order to avoid loss of confidence in the policy development process if new data/techniques become available and signal the need for future changes.

- **Comprehensible to stakeholders:** In many cases important and influential stakeholders involved in urban water management decisions do not have a technical background either in engineering or in resource planning. Professional hydrogeological expertise in city water management is generally absent and municipal water supply utilities may be more focused on the day-to-day operational needs of the present system, even where groundwater is a major urban resource. This was the case in both Narayanganj and Bishkek, where urban water management decisions did not appear to involve resource-knowledgeable institutions. Thus, while the underlying rationale may be subtle and the technical background complex, urban water management discussion documents need to be simple, clear and concise enough to engage municipal decision-makers with a minimum of technical jargon.

Precursor stages to the stakeholder consultation process

These three design criteria were employed not only at stakeholder consultation stage but also at the preceding pollution risk assessment stage, where the principal aim was to produce a single map that could be used by municipal planners and decision-makers (the Groundwater Resource Planning Map or GRPM). Precursors to the GRPM were intended to inform and act as reference material for water resources specialists and included a succession of maps (Figure 1): groundwater vulnerability map or GVM, potentially hazardous activities map or PHAM, superimposed GVM and PHAM or 'hotspot' map (see Morris et al., 2001 for a fuller description of this process).

The production of the maps served two purposes:

- concise provision of much background hydrogeological and environmental information in a comprehensible format available for reference if required to stakeholders, who may be technically-oriented but would be unlikely to be drawn from hydrogeologically-related disciplines;

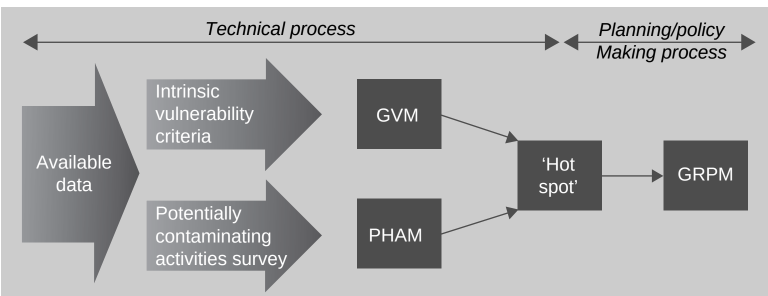

Figure 1. Evolution of component parts of a Groundwater Resource-Planning Map (GRPM).

- informing the project team's conceptualization of how the aquifer system was likely to respond to urbanization pressures, especially those likely to result in deterioration of raw water quality.

The maps were complemented by profiles of each city's groundwater setting and the corresponding water infrastructure using an urban questionnaire tool (Calow et al., 1999). Extracts from these are provided in the next section. It was notable, however, that the Groundwater Resource Planning Map itself did not evolve until policy formulation and stakeholder consultation had been almost completed, because that exercise guided what each city's map would emphasize.

CASE-STUDY CITY PROFILES

Narayanganj, Bangladesh

Physical setting: (from Morris et al., 2000) Narayanganj is a small city of about 1 million population, located 20 km south-east of Dhaka on the flat Ganges-Brahmaputra-Megna alluvial plain of central Bangladesh. A long-established industrial centre for the jute and hosiery industries, Narayanganj's proximity to Dhaka has favoured the recent development of light industry and it is now a national textile manufacturing centre, with factories undertaking all stages of production from spinning, dyeing/bleaching and weaving through to the making of garments and other finished cloth products. Other industries include soap-making, metal re-rolling and metal and wood furniture manufacture. The rapid and unchecked growth of Dhaka into a mega-city of 10 million inhabitants has seen inexorable encroachment on the rural hinterland west of Narayanganj and the city is likely in the mid-term to become an industrial satellite suburb of Dhaka. It had itself a high estimated annual growth rate of 5.8 per cent per annum during the 1990s.

Hydrogeological setting: Narayanganj is underlaid by an unconsolidated alluvial aquifer system of Quaternary age which is many hundreds of metres thick across the entire project area but in which only the top 250 m (and principally the top 150 m) is utilized for groundwater supply purposes. Complex lateral interdigitation of medium to coarse sands occurs with finer-grained sands, silts and clays. As a first approximation the system is considered to comprise an upper aquitard covering a shallow aquifer, which is separated from a deeper more productive aquifer by a lower, much thicker aquitard (Figure 2).

Vertical connectivity is likely to be variable, depending on thickness and the frequency of occurrence of fine-grained strata at any given location, and it is probable that there is hydraulic connection with the River Sitalakhya, whose channel is deep enough to incise into the upper aquifer sequence.

Bishkek, Kyrghyzstan

Physical setting: (from O'Dochartaigh et al., 2000) Bishkek has a population of approximately 800,000 and lies on the northern flanks of the Alatau range of the Tien

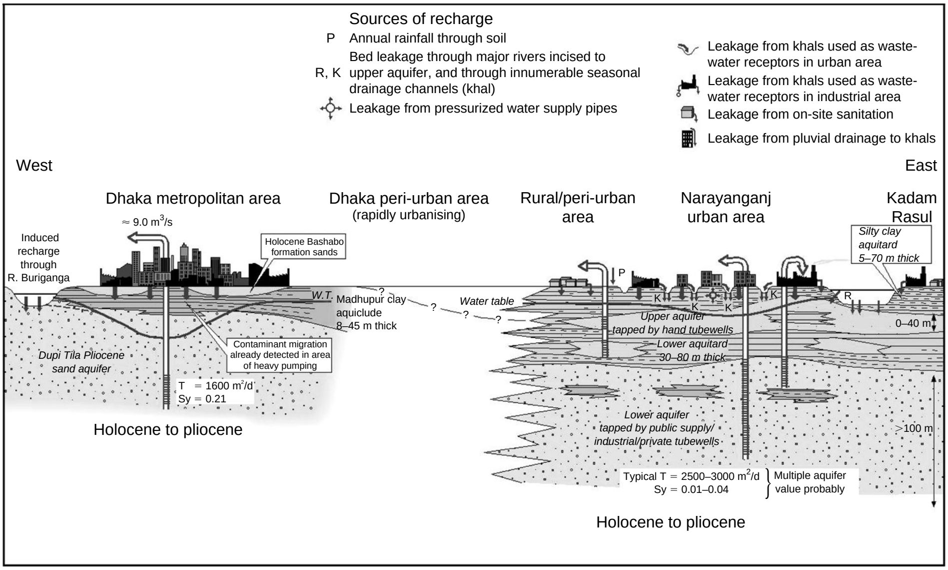

Figure 2. Groundwater setting of Narayanganj, Bangladesh.

Shan mountains in northern Kyrghyzstan. It is the country's capital and industrial centre and has witnessed changes since independence, notably the decline of the once-dominant Soviet military-industrial sector, and the increase in small private businesses, often with foreign investment. The city is 100 per cent aquifer dependent for potable, domestic, commercial and industrial water supplies which are provided by both intraurban and peri-urban wellfields.

Hydrogeological setting: The city's groundwater setting is hydrogeologically complex with a laterally heterogeneous fluvioglacial/alluvial multi-aquifer system of Quaternary age, which is in excess of 350 m thick in northern districts of the city. There is strong lateral and vertical variability, but as a first approximation the system fines laterally northwards away from coarse clastic piedmont deposits composed of coalesced alluvial fans fronting the foothills into more stratified deep alluvial plain sediments over a distance of less than 10 km (Figure 3).

Despite the semi-arid climate, there are extensive opportunities for recharge from snow-melt rivers and associated canal systems, draining the nearby Alatau range. Hydraulic connection with surface flow is thought to be strong across the southern piedmont area where the aquifer system is both unconfined and considered to possess strong vertical connectivity. More complex semi-confined conditions are present in the northern part of the city where three aquifer systems have been identified by a resource investigation project. Scope for significant pumping-induced vertical leakage exists, especially in the southern parts of Bishkek where low permeability horizons in the alluvial tract are thinner and less numerous. Unconsolidated sediments provide inter-granular flow conditions and the coarse alluvial and fluvio-glacial deposits comprising the aquifers have high transmissivities and significant vertical permeabilities, so urban boreholes abstract water at widely different depths.

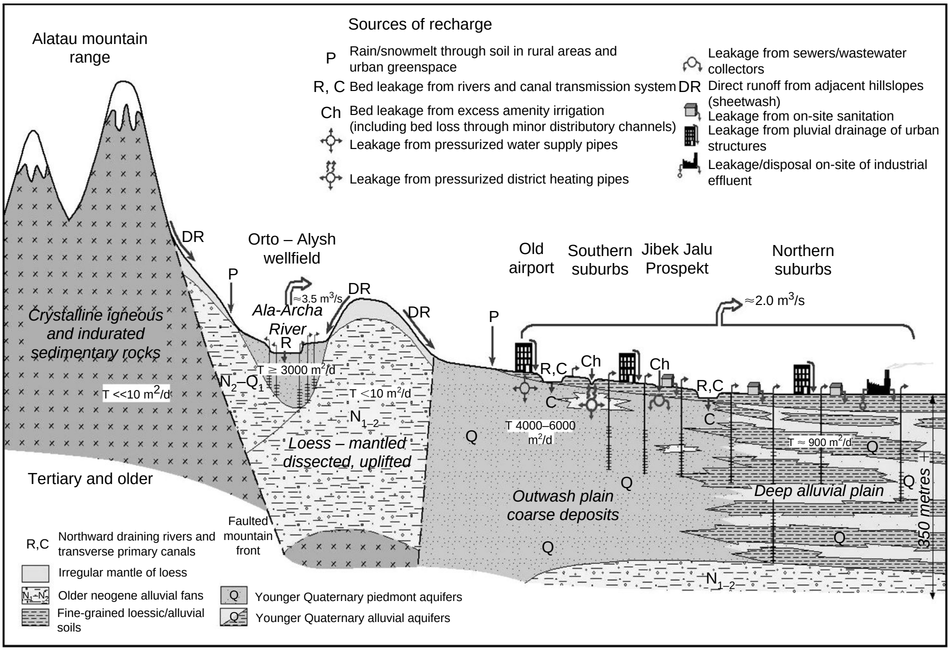

Figure 3. Groundwater setting of Bishkek, Kyrghyzstan.

IDENTIFICATION OF STAKEHOLDERS

Understanding development settings

Key to the identification of stakeholders in Bishkek and Narayanganj was a clear understanding of each city's groundwater development setting and urban water infrastructure. These were quite different in each city.

Groundwater development setting of Narayanganj: Groundwater provides more than 90 per cent of drinking water supplies in the study area and there is a similar high dependence for industrial and commercial needs. Large-scale groundwater abstraction for public supply and industrial use is mainly from the lower aquifer and located mainly within the urban area of Narayanganj. Broadly similar designs are employed for public supply and private industrial/commercial-use wells alike, so the deeper aquifer horizons are not reserved for potable use. The piped water supply does not extend beyond central Narayanganj and the per capita supply from the water utility to the urban part of the project area is estimated to average less than 45 l/p/d. Actual per capita usage is therefore almost certain to be widely supplemented with private supplies from either the shallow or (less frequently) the deep aquifer.

While the groundwater productivity in the shallow aquifer is considered too low for large abstractions, it is tapped by numerous narrow diameter boreholes equipped with hand pumps for drinking water and domestic supply purposes. These are locally known as No. 6 pumps. The operation of all such wells is the responsibility of local users, who could comprise just one family or a whole community. The total volume abstracted is unknown, but there seems little doubt that the upper aquifer is the primary source of potable supply for the rural and peri-urban population of the project area, as well as a supplement for urban households.

The resultant supply network is therefore diffuse, with piped water-supply coverage within much of urban Narayanganj but numerous hand pump-equipped shallow boreholes in rural and peri-urban districts. A large number of private and industrial wells exist. There is a register, but no published estimates exist of abstraction from the lower/main aquifer and it is very likely that there are many more unlicensed industrial wells.

There is no modern waste water and sewerage disposal system in the study area and dispersed on-site sanitation is widespread in urban, peri-urban and rural areas alike. Opportunistic use is made of the storm drainage system in central Narayanganj, mainly for sullage, but illegal foul-water connections are said to be common. There is no waste water treatment plant in the study area. Inadequate drainage results in frequent water-logging of many parts of the town and this has become a problem.

Groundwater development setting, Bishkek: There is a very extensive piped water infrastructure (pressurized hot water as well as drinking water mains, plus piped sewerage), but widespread on-site sanitation is practised in single/two-storey residential areas. Significant amenity irrigation of communal parts of residential areas occurs, using both canalized surface water and pumped groundwater. A highly productive but very localized peri-urban valley-fill well field, located 8 km south of the urban area, provides about two-thirds of the city's water demand, the balance coming from boreholes of various depths distributed throughout the city. These urban wells are screened extensively in the middle aquifer (typically >120 m intake depth), but the lower part of the upper aquifer (40 m–120 m) is also widely tapped.

The majority of abstraction boreholes are operated by the municipal water supply agency, which may provide water for both domestic and industrial processes, and there are also three separate reticulated systems for domestic water. One supplies cold water (domestic potable use), one supplies hot water taps (domestic non-potable use) and one supplies hot water for radiators (non-potable district heating use), the last of which appears to be a closed (non-consumptive) system, which is operated only during the winter. All come under the description 'public water supply'. The private urban water use categories are less important both in number and volume of water pumped; a small number of factories have private wells for potable or non-sensitive water supplies and there are also small numbers of private domestic and public municipal irrigation wells. Owner-operated supplies for commercial premises, hospitals and large state administrative buildings appear to be insignificant.

The waste water disposal system comprises a piped sewerage element to which industrial, commercial, apartment and public buildings, together with some low-rise residential housing, are connected and a dispersed on-site sanitation element in many low-rise residential areas. The relative importance and geographical extent of the latter is not well documented, but may be significant. A waste water treatment plant receiving domestic and industrial sewered effluent is located on the northern fringes of the city.

City stakeholder features

Thus in Narayanganj:

- industrial users are important and influential stakeholders, meriting extra effort in consultation;

- there is overlap between rural and urban water supply agencies, especially in the peri-urban area;
- users of the shallow aquifer, which still serves as a resource as well as a receptor, will be difficult to represent;
- no primary stakeholder groups were identified.

While in Bishkek:

- state-sector agencies remain the predominant stakeholder group members;
- only secondary stakeholders could be identified;
- a post-independence depression in industrial activity offers opportunities for context-sensitive planning intervention to support a sound basic infrastructure;
- even though the upper aquifer is not widely used for potable supply, likely high vertical permeabilities, especially in the southern half of the city will favour rapid vertical movement of pollutants towards much deeper aquifers apparently remote from the land surface.

In both cities it proved impossible to identify representative primary stakeholders (those with a direct resource interest, including groundwater users) who could participate in a consultation process. Although participation by primary stakeholders is looked for, the degree of organization (and thus representativeness) of such user groups may only occur in some urban contexts and may therefore be merely desirable rather than indispensable. In Narayanganj the absence of this stakeholder class was more than compensated for by the diversity of secondary stakeholders (intermediaries in the delivering of policies, projects and services to primary stakeholders). These were drawn not only from public sector agencies/ministries but also local government and trade/industry associations. In Bishkek public sector organizations dominate the stakeholder spectrum, but much underfunding and quite poor coordination post-independence has fostered a diversity of views from the agencies involved, ensuring active discussion of options.

THE STAKEHOLDER CONSULTATION PROCESS

Once identified, stakeholders were engaged through the medium of a periodic bilingual newsletter (Russian/English, Bangla/English), of which about half a dozen were issued in each city over an 18-month period leading up to a stakeholder workshop. The newsletters were kept short and focused, each edition communicating one aspect of the policy consultation process in a logical sequence leading through problem recognition, identification of future threats and consequences, setting of remedial/preventive objectives and targets, and into policy formulation (Figure 4).

Thus different editions:

- communicated a synopsis of results from the first part of the project, including the 'hot-spot' (potentially contaminating industries on vulnerable aquifer) map;
- presented Strengths, Weaknesses, Opportunities, Threats (SWOT) analyses (Figure 5, Table 1) of different facets of the city water infrastructure (public and private water supply, waste water and solid waste disposal, management and regulatory control, planning). This analysis method was employed as it lends itself to brevity in presentation and is likely to be already familiar to some stakeholders from its wide use as a commercial business/market analysis tool;

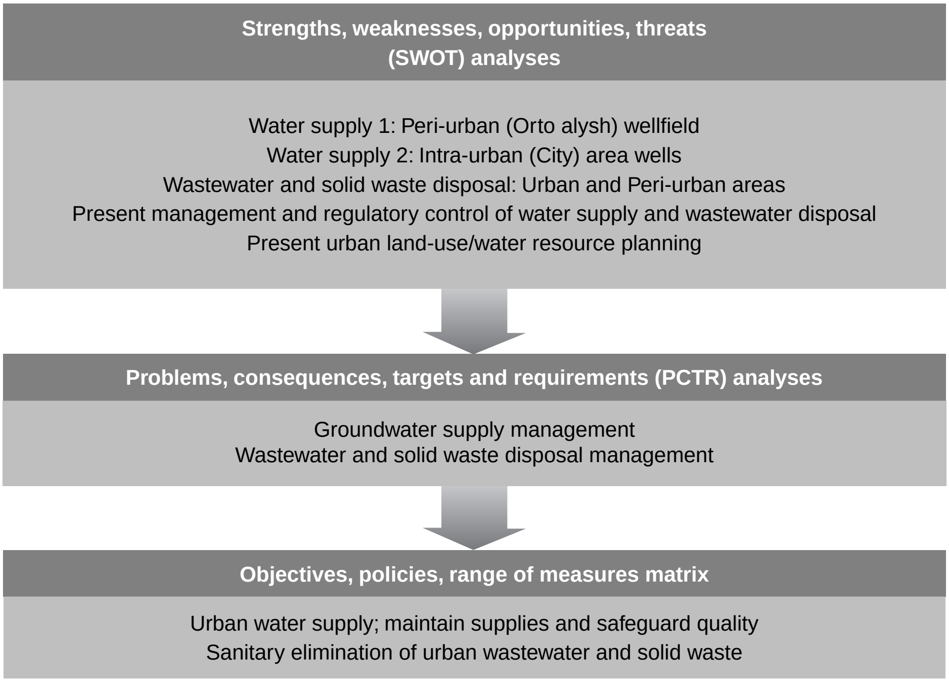

Figure 4. Evolution of groundwater protection policies for a groundwater dependent city.

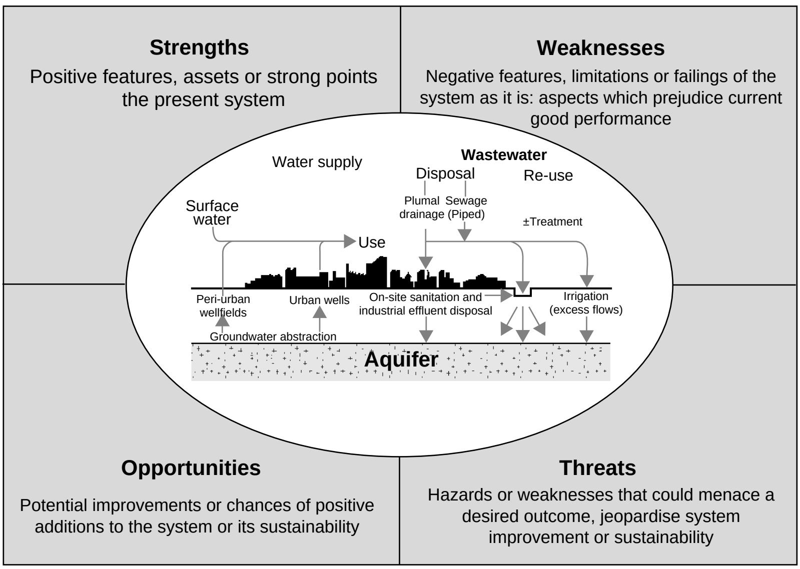

Figure 5. Strengths, Weaknesses, Opportunities, Threats (SWOT) analysis for groundwater-dependent city.

Table 1. Example from Bishkek of SWOT analysis of one sector of city's water infrastructure.

Water-supply infrastructure – peri-urban (orto alysh) wellfield	
Strengths 1. Located up groundwater gradient outside city and at topographically higher elevation 2. High productivity in small area due to very high permeability, high specific yield and river-aquifer leakage 3. Small interference effects between wells due to high permeability; permits small, compact wellfield which simplifies distribution and treatment infrastructure 4. Wellhead protection/sanitary control area under control of utility 5. Wellfield catchment areas still predominantly rural with few roads 6. Aquifer baseline water quality likely to be high: significant quality derogation tolerable.	*Weaknesses* 1. No control over catchment outside wellhead protection/sanitary control zones, despite existence of control directives 2. Wellfield is susceptible to various pollution threats: aquifer is high vulnerability zone, adjacent to area of extreme vulnerability (losing reach of Ala Archa river) 3. Well catchments likely to include significant river leakage element; this could act as short-cut for pollutants in surface water to penetrate down to borehole intake levels 4. Wellfield composite catchment not commonly known; time-of-travel zones not defined 5. No coordination of either quantity or quality surveillance monitoring of piedmont recharge zone.
Opportunities 1. Area of wellfield is compact and therefore easy to define for regulatory protection purposes 2. Wellfield joint catchment would be relatively easy to define by existing groundwater modelling techniques 3. Scope for progressively stricter control measures using time-of-travel zones defined by standard modelling techniques 4. Proximity to National Park area could make regulation of potentially polluting activities easier for public to accept 5. Stakeholder/public understanding of water supply protection issues could be improved by easier access to key urban water trends, e.g. by means of a public information website.	*Threats* 1. Risk of microbial pollution from on-site sanitation of summer house cooperatives either via river leakage or direct to aquifer: moderate probability hazard which could, however, be managed by treatment (raw water disinfection) 2. Susceptible to pollution incidents, e.g. spillage from traffic accidents: low probability hazard but with potentially serious consequences 3. Major pollution incident affecting wellfield would threaten 70 per cent of city's water supply, only part of which could be offset by use of reserve wells in city 4. Adjacent sensitive central government facilities may increase risk of terrorist attacks on infrastructure due to proximity 5. Hazard of water quality deterioration due to use of agrochemicals for livestock pest control and in cultivated area, and increasing density of domestic properties in catchment 6. Risk of flood/mudflow damage to wellfield installations due to proximity of wellfield to axis of narrow valley.

Table 2. Example from Narayanganj of policy options developed by stakeholder consultation for aquifer protection (water supply matrix).

Objectives	Urban water management policies	Examples of policy measures to achieve targets — Degree of intervention: Weak → Moderate	Moderate → Strong — *Policies in left-hand column plus:*
Maintain groundwater supplies	1. Monitor groundwater levels and use results to manage urban aquifer	Install/maintain a water level monitoring system	Use results in directing/controlling abstraction
	2. Gain better understanding of groundwater resource so as to develop it in a sustainable way	Use model to update vulnerability assessment/provide source catchments	Undertake comprehensive 'master plan'-type water resource study, including improvement of groundwater flow model
	3. Value groundwater realistically to aid management of finite natural resource	Introduce pricing policies for public water supply to recover present supply costs	Extend pricing policies for public water supply to cover expansion to meet increasing demand ± future additional treatment requirement (i.e. long-run marginal costs)
	4. Mitigate effects of increasing abstraction on shallow aquifer users	Provide guidance to public on well interference effects	Provide substitution supplies where shallow aquifer likely to suffer early dewatering
	5. Constrain groundwater level decline in lower aquifer	Enforce well spacing criteria to minimize interference effects between neighbouring deep wells	Reserve good-quality water for potable/sensitive use and encourage use of shallow (poorer-quality) groundwater for non-sensitive uses (by abstraction licensing/quota system, licensing of drilling companies, amending building regulations to include water conservation technologies, subsidize water-efficient technologies/appliances
	6. Privatization of specific functions	Privatize meter reading and bill collection to ensure recovery of cost of supply	Devise water-charging system and implement collection sanctions that are both fair and transparent, and support measures that will enhance supply cost recovery

Safeguard water quality	7. Monitor groundwater quality and use results to manage urban aquifer	Install/maintain a water quality monitoring system	Use results in directing/controlling abstraction
	8. Eliminate shallow aquifer pollution due to poor well construction/ maintenance	Public awareness/schools education programme on wellhead maintenance	Proactive wellhead maintenance teams
		Provide well construction guidelines and make compliance mandatory	Training/quality assurance of drilling teams
Improve management	9. Encourage economic expansion consistent with conservation of urban groundwater resources	Classify industrial/municipal activities into hazard classes	Identify different hazard class suitabilities: encourage compliance by use of economic/ regulatory instruments such as building controls
	10. Support integrated pollution control for all existing industries	Use hazard + vulnerability classifications to identify industries of posing threat to groundwater	Prioritize industries posing main threat for integrated pollution control measures or relocation
	11. Enhance professional capacity	Employ professionals in utility to manage aquifer for its prime water supply function	Employ professionals in utility in key area of hydrogeology and groundwater resource development
	12. Implement existing laws and regulations to achieve national policy goals	No lesser-scale intervention effective, see right-hand column	Central government (executive and legislature) to explicitly support municipal and utility efforts to enforce existing pollution and abstraction control regulations
	13. Encourage people participation and public awareness.	Educate press in groundwater issues to better inform public.	Formalize stakeholder forum and act on consenus decisions recommended by them.

- provided Problems, Consequences, Targets, Requirements (PCTR) tables for water supply and waste water/solid waste management. This analysis was intended to bring together public and private uses in such a way as to show the commonality of issues, the resolution of most of which would at some point require either the active participation (or at least the absence of opposition) of private sector users;
- suggested a range of policy options based on broad strategic objectives (maintaining groundwater supply, safeguarding water quality, sanitary elimination of urban waste water/solid waste). These were ranked in terms of the degree of intervention needed to put the policy into effect (Weak to Moderate to Strong), with an implied link to relative efficacy (e.g. Table 2);
- reported the results of the stakeholder workshop, at which each stage of the policy consultation process was revisited and reopened for modification by consensus and where the policy options were debated, revised and/or added to.

OBSERVATIONS ON THE STAKEHOLDER CONSULTATION PROCESS

1. The workshop comprised the penultimate stage of the consultation exercise in each city because the research project was coming to an end. However, in other programmes it would be expected that a successful workshop would lead to the establishment of a stakeholder forum, usually facilitated by sponsorship of an important stakeholder such as the city water utility, municipal planning department or public health agency. Unsolicited, participants at both workshops identified this as a recommendation. The direction such a forum might take would, of course, vary with the energy and influence of the participant individuals and the degree of autonomy enjoyed by the municipality in terms of planning regulation.
2. In a few cases there might be enough impetus generated to enact municipal ordinances, the enforcement of which would be much assisted by the prior consensus developed by a representative stakeholder forum. More typically it might become a lobby, seeking to influence central government or a particular ministry into the enforcement of existing environmental/water resource regulations or the enactment of new enabling legislation.
3. In Narayanganj participants identified a need for more involvement at local level in the planning process. Given the city's proximity to rapidly-expanding Dhaka, this reflected a general uneasiness over the remoteness and lack of transparency of the metropolitan planning authority, which currently handles planning issues in the region around the capital city. In Bishkek it was felt that there was scope in the future for directing water supply and sanitation development assistance funds into urban infrastructure development/aquifer protection instead of concentrating exclusively on the rural sector, as at present.
4. It seems inescapable that if groundwater protection is to be brought into the municipal planning process, then stakeholder policy forums will have to enter the political arena if resultant planning regulations are to be enforceable and enforced.
5. Stakeholders liked the general approach of working openly through the policy development process, although this only became apparent during the workshops since feedback from stakeholders receiving the newsletters beforehand had been very poor. Nevertheless, the newsletters provided a means to drip-feed quite complex information

which could never be assimilated in the time available to a workshop audience, while the workshops provided the enabling forum for frank discussion of problems. Our view is that structured newsletters and workshops complement each other and proved together to be effective aids to stakeholder consultation.

6. There is nonetheless a paradox in the use of workshops for stakeholder consultation purposes. The meetings would be most influential and high profile if the decision-makers within each stakeholder group attended them. Yet the time required for a workshop means that staff detailed to attend are rarely the most senior members of each stakeholder group. So secondary stakeholder consultations are burdened not only with ensuring that meeting deliberations are transmitted effectively to agency decision makers, (perhaps several levels higher in the organizational hierarchy) but also with the inability of participants to speak authoritatively on behalf of their respective agencies. This prolongs the consultation exercise with, inevitably, the risk of loss of credibility and interest in the process.
7. At both workshops comments by participants showed that although only those concerned with the city's water infrastructure attended, such cross-sectoral involvement was unknown. Even allowing for the novelty of the approach, it was clearly welcomed by attendees as a positive contribution to the urban water development process. There may be lessons to be learnt here on institutional involvement for international development agencies involved in urban water infrastructural improvements in small to medium-size cities.

CONCLUSIONS

- Despite a limited resource and knowledge base, locally appropriate groundwater protection plans are being developed in the case-study cities.
- Stakeholder consultation contributed significantly to the development of policy options for aquifer protection in both cities.

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NOTES

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